University of Alberta

Woodland Caribou-Wildfire Relationships in Northern Alberta

by

Jesse Scott Dunford

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of *Master of Science*

in

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ABSTRACT

For the past several millennia, wildfires have been the largest disturbance agent in the northern Alberta's boreal forest. Wildlife species such as woodland caribou (*Rangifer tarandus caribou*) have existed and evolved with these disturbances. In more recent history though, populations of caribou in Alberta have declined, and knowledge of how wildfires influence caribou habitat use and their food resources has become an important component of the efforts to stem population declines. Wildfires are a significant disturbance element for caribou because they incinerate slow-growing lichens that caribou depend on for winter forage. I studied the relationship between woodland caribou and wildfires in northern Alberta to determine how such changes in forage resources influence caribou. This work has three foci: (1) to document how wildfires change lichen availability for caribou, (2) to address how caribou distribution changes with the occurrence of wildfires, and (3), to measure how caribou use and select habitats that have burned. I then use these data, together with measures of caribou responses to anthropogenic disturbances, to explore what caribou population trends may look like over the next century, given different scenarios of natural and anthropogenic disturbance.

I found that in preferred caribou habitats (ombotrophic bogs), lichen abundance is relatively low, and following wildfires, lichen regrows to maximum levels within 40 years. Drier sites appear to contribute to faster growing and more abundant lichens. The loss of lichens from wildfires had little relationship with the large-scale distribution patterns of caribou, as caribou reuse of annual ranges was independent of

the occurrence of wildfire. Caribou exposed to wildfires also showed no difference in annual home range size before and after wildfires. Within annual ranges, caribou generally avoided young seral stage habitats and selected older areas. Seasonality in selection and use patterns was not apparent. Habitat use also indicated that selection of burned areas by caribou may vary with the extent of wildfire, and avoidance of burns decreased with increasing abundance of wildfire. Using these measures of caribou habitat use, and the current growth of industrial disturbances, simulation modelling predicted a reduction in caribou populations of 98% within 50 years; however, natural disturbances played a limited role in population trends.

Wildfires in northern Alberta certainly reduce the availability of lichens for caribou, and can result in detrimental, but non-permanent, habitat alteration. Yet distribution, fidelity, and habitat selection of caribou in Alberta suggest that although caribou are likely exposed to habitats of different ages, they are able to meet their lichen requirements in patches of older areas within younger stands (i.e., unburned islands within fires) or in other parts of their range. I suggest that the best course of action for conserving caribou in Alberta lies not with effective firefighting or wildfire prevention, but the adoption of improved industrial operating techniques, aggressive reclamation of existing footprints, and a temperance for development in caribou range.

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CHAPTER 1. GENERAL INTRODUCTION AND THESIS OVERVIEW

Background and Rationale

This work focuses on documenting the relationship between woodland caribou (*Rangifer tarandus caribou*) and large-scale wildfire events in the boreal forest of northern Alberta. Interest in this project began because caribou are threatened in Alberta (Dzus 2001, COSEWIC 2002), and evaluating how natural disturbances influence caribou and their food resources is key to understanding caribou habitat use and distribution. Additionally, Alberta has faced, and will continue to face in the future, multiple land uses and habitat alterations. Anthropogenic habitat alterations, often termed 'footprints,' are often co-occurring, meaning that disturbances can no longer be examined in isolation, especially in context of their influence on wide-ranging species such as woodland caribou (Schneider 2002). Here I measure the influence of natural disturbance on woodland caribou and their winter food, and incorporate those measures with measures of other disturbance elements to explore the possible habitat availabilities for Alberta caribou in the future.

Wildfires in boreal forests

The largest single agent of disturbance in boreal ecosystems comes not from anthropogenic activities such as forest harvesting or oil and gas exploration, but from the incineration of forest stands due to wildfires. Successional stages of plant communities are often reset by the occurrence of wildfires, as older trees and

vegetation are replaced by new growth (Johnson 1992). The probability of wildfires occurring is influenced by weather patterns and local fire fuel build-up (Newark 1975, Flannigan and Harrington 1988, Johnson and Wowchuk 1993). Although the cycle of wildfires in boreal regions (the time required to burn a given area equal in size to a given study area [Johnson and Van Wagner 1985]) may have changed in the past century due to climate changes and fire suppression activities (Johnson et al. 2001, Ward et al. 2001), long-term averages indicate that between 1% and 2.5% of forested areas burn annually in western Canadian forests (Murphy 1985, Cumming 1997). Fires greater than 200 hectares are responsible for the vast majority of area burned, while smaller fires are far more numerous. The predominance of very rare but large fire events means that the probability of wildfires burning any given area can best be modeled by a serially random and independent draw from a lognormal distribution (Armstrong 1999).

The abundance of wildfires in boreal regions has meant that animal species have co-evolved with natural disturbance over the past several thousand years in North America. Because wildfires restart successional pathways for plant communities, the forage resources for wildfire species can be significantly altered by the presence or absence of wildfires, and many animal species will use burned or unburned areas in response to such resource availability. Moose (*Alces alces*) for example, are known to favour browse rich in nutrients that regrows following fires (Peek 1974, MacCracken and Viereck 1990), while woodpeckers are often found amongst recently burned trees searching for insects found in decaying wood (Hutto 1995; Murphy and Lenhausen

1998). The continued succession and termination of plant and forest communities that results from wildfires is assumed to be beneficial for wildlife species in general, because the process serves to maintain the diversity and heterogeneity of habitat types (Schaefer and Pruitt Jr. 1991). However, for species that depend on the late-succession vegetation, the impact of large-scale wildfires can quickly reduce the availability of certain forage resources.

Caribou-wildfire relationships

Caribou are lichen specialists in winter, so the higher lichen biomass in late successional forests enables caribou to meet winter forage needs (Darby and Pruitt Jr. 1984, Schaefer and Pruitt Jr. 1991, Thomas et al. 1994, Edmonds and Bloomfield 1984). Thus, woodland caribou are known to occupy areas that have failed to burn for long periods of time, often termed 'old-growth habitat.' Similarly, caribou may avoid recently burned areas because of a lack of lichens, although this has only been well documented amongst caribou with seasonal winter and summer ranges (e.g. Thomas et al. 1996, Schaefer and Pruitt Jr. 1991). In such populations, the ability of caribou to occupy alternate areas unburned by wildfires was seen as key for caribou to cope with such natural disturbances, reinforcing a premise that population dynamics are likely not influenced by the occurrence of wildfires (Bergerud 1974). However, the response of wide-ranging and non-migratory caribou in Alberta to wildfires was unknown.

Cumulative Effects Assessment

While measures of disturbances may take place independently, understanding the cumulative influences of disturbance, both natural and anthropogenic, is seen as the best course from which to build a management strategy for conservation. Natural disturbances such as forest fires that alter caribou food resources may be overlooked in relating disturbance to wildlife distribution and behaviour, but in northern Alberta, wildfires disturb far more area than forestry, oil and gas exploration, or mining activity. To account for these multiple disturbances, I used ALCES ® (A Landscape Cumulative Effects Simulator; Forem Technologies) to model how caribou populations and habitat availability may behave under different scenarios of development and climate change. The advantage of this approach is that ALCES ® accounts for the cumulative footprint of disturbances, both natural and anthropogenic, to allow strategic assessments of the best course(s) of action for caribou conservation.

Thesis Overview

This work is arranged to address the question of how a change in forage resources resulting from wildfires influences woodland caribou in northern Alberta. I document the availability of lichens within preferred caribou habitats and how lichen availability varies with time since disturbance (Chapter 2), how caribou respond to forage changes in a short-term manner through changes in distribution (Chapter 3), and in a long-term manner through habitat selection (Chapter 4). I then integrate my findings with other works of caribou-disturbance relationships to compare how

caribou may fare among possible landscapes of Alberta over the next 100 years (Chapter 5). A more detailed outline follows:

Chapter 1: This introductory chapter briefly reviews the role of wildfires in boreal systems, and summaries previous works on caribou-wildfire relationships. I also speak to the importance of cumulative effects assessment in woodland caribou management.

Chapter 2: Caribou rely on slow-growing lichens as a source of energy in winter; hence, it is assumed that caribou responses to wildfire are due to the loss of lichen availability. In Alberta peatlands though, the abundance of lichens has not been documented, so this assessment will include a comparison of the lichen abundance in peatlands among stands of various ages. I document the post-fire regrowth of lichens and other habitat attributes in treed bogs, the biomass of terrestrial lichens, and the factors influencing abundance and the growth rate of lichens.

Chapter 3: This chapter examines changes in caribou distribution following a large wildfire event, a measure of the immediate response of caribou to winter forage loss. I test whether caribou distribution is altered beyond that which was seen in years without disturbances. I also measure how range fidelity and annual range size is influenced by wildfire, and explore whether range quality and the amount annual ranges are burned is related to range fidelity.

Chapter 4: This chapter presents a more long-term measure of caribou responses to changing forage conditions. Here I document the habitat use and selection patterns of caribou in four regions of Alberta; I determine what seral stages of habitats caribou use and select, and the extent of such use and selection. I compare seasonal selection patterns and selection between peatlands and uplands, and assess whether habitat selection is a means for caribou to cope with varying degrees of forage losses from wildfire.

Chapter 5: Here I use my estimates of forage regrowth and caribou responses to wildfires from previous chapters to parameterize ALCES ®, and along with information from other caribou studies, assess the ability of caribou to cope with the cumulative effects of development and climate change. The role of climate change was explored in scenarios because of the potential for climate changes to alter parameters of wildfire cycles. I use ALCES ® to predict the availability of caribou habitat and the trend in a hypothetical caribou population over the next 100 years with three development scenarios (none, current trends, and best practices) under two climate scenarios (current and climate warming). The goal of this work was to make strategic comparisons of management scenarios, rather than conduct a Population Viability Analysis (PVA) or predict habitat availability with precision.

Chapter 6: This final chapter presents general conclusions on caribou-wildfire relationships in northern Alberta, and discusses management strategies that may be effective for caribou conservation in this environment.

NB. Chapters 2 through 5 are in manuscript format, so some redundancy exists in descriptions of the study area and methods, and are written in first-person plural because of the contribution of multiple authors.

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CHAPTER 2. LICHEN SUCCESSION AFTER WILDFIRE IN PEATLANDS: IMPLICATIONS FOR WOODLAND CARIBOU

Introduction

In boreal forest ecosystems, wildfires are the dominant disturbance agent at the landscape scale (Johnson 1992), and the succession of plant communities following wildfires influences the use of landscapes by animals associated with the resources of particular seral stages. For example, Black-backed Woodpeckers (*Picoides arcticus*) are often found in early successional stage forests because of the high biomass of insects associated with decaying wood (Hutto 1995; Murphy and Lenhausen 1998), while moose (*Alces alces*) are associated with early successional forests because of an abundance of shrubs and forbs (Peek 1974; MacCracken and Viereck 1990). Conversely, woodland caribou (*Rangifer tarandus caribou*) are thought to use late succession forests because higher lichen biomass in older forests is necessary for caribou to meet forage requirements (Klein 1982; Thomas, Edmonds and Brown 1996, Edmonds and Bloomfield 1984, Schaefer and Pruitt 1991).

Woodland caribou have been shown to use lichens – almost exclusively in many cases – as their source of winter forage (Scotter 1967; Boertje 1984; Thomas et al. 1996). While lichens contain little protein (Scotter 1965a; Pulliainen 1971), they contain a reliable source of digestible energy (Cameron 1972, Pearson et al. 1975). Lichens of the genus *Cladina spp.* and *Cladonia spp.* are selected and used by woodland caribou at foraging sites (Edmonds and Bloomfield 1984; Bradshaw et al. 1995; Johnson et al. 2001). In many regions, woodland caribou are able to use arboreal

lichens when terrestrial lichens are inaccessible because of snow conditions (Rominger and Oldemeyer 1990; Rominger et al. 1996). However, terrestrial lichens in boreal Alberta are far more common, and snow depths rarely make them inaccessible for caribou.

In northern Alberta, caribou are found primarily in peatlands (treed bogs and fens; Bradshaw et al. 1995, Anderson 1999, Schneider et al. 2000) and range over 500 km^2 annually (Bradshaw et al. 1995; Stuart-smith et al. 1997; Schneider et al. 2000). Bradshaw et al. (1995) demonstrated that forested bogs were selected over all other habitat types for winter-feeding sites, reasoning that the xeric substrate of raised bogs coincided with greater lichen biomass. More recent work indicates that early seral stage peatlands can comprise a substantial component of caribou home ranges (Dunford 2003). The availability of lichens in these regions has not been previously documented, nor is it known how lichen resources change following wildfires in peatlands.

Because *Cladonia* and *Cladina* lichens grow very slowly (Thomas et al. 1996; Grosvernier et al. 1997), lichen biomass is thought to be a function of time since disturbance (predominately wildfire in boreal systems). Lichen abundance in peatland regions, however, may be influenced by the abundance of *Sphagnum*, in addition to time since wildfire. Peatlands are dominated by *Sphagnum* mosses (Vitt et al. 1996) that continually grow and accumulate, leading to the phenomenon of paludification, whereby the accumulation of organic matter results in the rising and spread of peatlands. Paludification and the presence of *Sphagnum* mosses have led to a system

where, at a very fine scale, the substrate for lichen growth is 'vertically dynamic' and contains little mineral soil (Camill 1999). The interaction of lichen and *Sphagnum* growth in peatlands is not well documented, and competitive or facilitative interactions between the consistently rising *Sphagnum* and terrestrial lichens may create microclimatic conditions that enhance or deter the abundance and growth of lichens.

Boreal populations of caribou are threatened in Alberta (Dzus 2001, COSEWIC 2002), and resources available to caribou in peatlands are a key component of the species' ecology. Because of the prevalence of wildfires in boreal systems, understanding how resource availability changes following disturbance is an also an important step in conservation planning for caribou (Dzus 2001; McLoughlin et al. 2003). The objectives of this work were to document (1) the abundance of terrestrial lichens in peatlands, (2) how terrestrial lichen abundance changed with post-wildfire succession in peatlands, and (3) determine which factors, in addition to time since disturbance, may contribute to the availability of lichens in peatlands. We then discuss the implications of our measures of lichen availability for caribou in northern Alberta.

Methods

Study Area

We surveyed peatlands throughout northern Alberta within known caribou home ranges (Figure 2.1). Elevation ranged between 500m and 1000 m. All regions were found within the boreal forest natural region of Alberta (Alberta Environment

2002). We limited site selection to forested ombotrophic bogs in an attempt to ensure consistency of peatland habitats compared. Forested bogs were generally dominated by species of *Sphagnum fuscum*, Labrador tea (*Ledum groenlandicum*), and small bog cranberry (*Vaccinium oxycoccus*). Tree species most often found were black spruce (*Picea mariana*) and tamarack (*Larix larcina*). Terrestrial lichens were almost exclusively *Cladina mitis,* although *C. rangiferina, C. stellaris,* and *Cetraria spp.* were also present.

Survey Methods

The Peatland Inventory of Alberta (Vitt et al. 1996) was overlain with digital wildfire data (Alberta Sustainable Resource Development 2002) to locate peatland study sites that had been disturbed by wildfire. The Peatland Inventory is based on homogeneous polygons of vegetation types identified from aerial photography. Each polygon within the Inventory is categorized according to a hierarchical classification system that reflects the percentage composition of different wetland types. We used a Geographic Information System (GIS; ArcView 3.1, Environmental Systems Research Institute Inc. 1998) to select polygons that (1) had >70% treed bogs, (2) fell within the boundaries of caribou home ranges, and (3) and contained wildfires that occurred since 1931. This technique established target areas from which we selected sites inside the fire edge that were accessible either with helicopter or ground transportation. Sites were confirmed to be treed bogs prior to sampling. We attempted to select a gradient of sites based on time since wildfire occurrence ($n = 48$), in addition to sites that had

no record of fire disturbance in the past 70 years, termed 'mature' sites (n = 25). In sites identified as burned within the past 70 years, we consciously attempted to exclude areas that appeared to be residual unburned patches, as evidenced by the lack of standing dead trees.

We established a 100 m long transect in a random direction at each site. At 10 m intervals along transects, we placed ten circular plots measuring 1.80 m in radius (10 m^2). Within each plot, we visually estimated percentage cover of lichens, forbs, shrubs, *Sphagnum* mosses, *Carex spp.*, and graminoids (predominately sheathed cotton-grass; *Eriophorum vaginatum vaginatum*). For each variable, we calculated the mean of the ten plots for that transect, and used those values as the sample unit in subsequent analysis. We also constructed four plots measuring 25 cm by 25 cm (0.0625 m^2) at two mature sites from which to derive a lichen biomass-percent cover relationship. We visually selected plots that appeared to contain a range of percent coverage of lichen. We photographed and collected all live vegetation within each plot, and separated out *C. mitis*, which we air-dried and weighed. Percent cover of *C. mitis* was then estimated from enlarged photographs.

In order to determine lichen growth rates and age, we also collected enough *C. mitis* to fill a 1 L plastic bag from three random locations along transects (n = 20 from mature sites, n = 26 from burned sites). From the aggregated sample at each transect, 25 individual strands of lichen were randomly selected. These were measured from the base of the live portion of the lichen to tip, and the number years of lichen growth

determined by counting the number of branches along the strand length (Vitt et al. 1988). We used the mean lichen growth rate for each transect in subsequent analysis.

Statistical Analysis

To determine the abundance of lichens in peatlands, biomass at each mature site was estimated from the equation of the regression line of dry weight biomass per 0.0625 m^2 on percent cover. We extrapolated lichen biomass from each transect mean to a per hectare unit for ease of interpretation.

We aimed to first model lichen abundance with time since disturbance, thus we present a polynomial regression of lichen cover on time since disturbance. Lichen cover was log-transformed to meet assumptions of normality in the regression. To explore other factors that influence the abundance of lichens in post-wildfire peatlands, we used a multiple regression of lichen cover on time since disturbance, cover of *Sphagnum*, forbs, shrubs, *Carex spp.*, and graminoids. Lichen, Sphagnum, and graminoid cover were also log-transformed to meet assumptions of normality in the regression. Only the variables significant at an alpha of 0.05 in the regression are presented below.

The recent growth rate of lichen (recent growth was termed to be that of the lichen strand which was not actively decaying) was determined by dividing the strand length by number of years of growth. We then regressed recent lichen growth rate on time since disturbance and again used a multiple regression of lichen cover on time since disturbance, cover of *Sphagnum*, forbs, shrubs, *Carex spp.*, and graminoids to

determine what factors influence the growth rate of lichens. Lichen, *Sphagnum*, and graminoid cover were also log-transformed to meet assumptions of normality in the regression. Again, only the variables significant at an alpha of 0.05 in the regression are presented below. All statistical analyses were conducted using SPSS version 10.1 (SPSS Inc. 2001).

Results

We derived a relationship between the percent cover of *C. mitis* and biomass per hectare to be $y = 30.475x$, where y represented the dry-weight biomass of *C. mitis* per hectare and x represented the percent cover of lichen per hectare (Figure 2.2). From this equation we estimated that in mature sites, mean terrestrial lichen biomass was 660 kg*hectare^{-1} (SE = 63.3, range 140 to 1258, $n = 25$). The mean number of years of live *C. mitis* growth above the substrate in mature sites was 12.33 (SE = 0.22, range 11.04 to 14.24, $n = 20$).

Lichen cover showed a strong quadratic relationship with time since disturbance (Figure 2.3). We estimated that lichen cover reached levels found in mature sites in as little as 30 years, based on the lower 95% confidence interval of our estimate of mean lichen cover in mature sites. Lichen cover had a positive relationship with time since disturbance ($P < 0.00001$) and a negative relationship with *Sphagnum spp.* cover ($P < 0.00001$; Figure 2.4), indicating that lichen abundance increased with age of sites, and was lesser with greater abundance of *Sphagnum* cover. The regression of lichen cover on time since disturbance and *Sphagnum spp.* cover explained

considerable variation in the model ($r^2 = 0.77$). Percent cover of other sites variables including forbs, shrubs, *Carex spp.*, and graminoids were non significant predictors of lichen cover in the regression ($P > 0.05$ for all variables).

The mean growth rate of lichen was 4.80mm*year^{-1} (SE = 0.12, range 3.69 to 5.49, $n = 20$ in mature sites). Growth rate of lichens was explained reasonably well by graminoid cover and time since disturbance ($r^2 = 0.35$), and was positively correlated with both time since disturbance ($P = 0.014$; Figure 2.5), and graminoid cover ($P = 0.049$; Figure 2.6), indicating that *C. mitis* tended to grow faster as bogs aged and in sites with greater abundance of graminoids. Residual plots (Figures 2.5 and 2.6) depict the relationship between the dependent variable and the independent variables controlling for other variables in the multiple regression.

Discussion

Lichen Abundance in Peatlands

The mean abundance of lichens in peatlands we measured is much lower than non-peatland systems, nearing an order of magnitude difference from some stands in Alaska (Thomas et al. 1996; Scotter 1965b, Courtright 1959). Our relatively low measure of lichen biomass likely reflects the moisture and substrate conditions that inhibit lichen abundance in peatlands. Most notably, the high water table in peatlands probably limits lichen production in this environment, contrary to drier pine and spruce stands that support greater lichen biomass. Additionally, we found only 12.33 years of lichen growth were above the peat substrate on average, so the acidic water

within peat substrates may decompose lichen strands in peatlands quickly, further contributing to the low biomass of lichens in peatlands.

Sphagnum mosses dominate peatlands, and at very small scales (i.e. <10 m^2) this substrate appears to reduce the abundance of lichen. The negative correlation of lichen cover with *Sphagnum* indicates that lichens may be less abundant in sites with greater *Sphagnum*, because *Sphagnum* presence is typical of wetter sites, in which lichen do poorly, or because Sphagnum growth is competitive, and faster growing *Sphagnum* overgrows the slow-growing lichens. At a larger scale though, bogs that contain more abundant lichens could results from the larger process of paludification; whereby the accumulation of peat over decades and centuries can lead to elevated bogs, with more xeric substrate conditions conducive to lichen growth and cover.

Lichen Regrowth Following Fire

Lichen abundance in surveyed bogs appeared to reach a maximum within 40 years post-fire, a relatively short period of time in comparison to other non-peatland systems caribou inhabit (Scotter 1965b; Thomas et al. 1996). The declining trend in lichen abundance from years 40 to 60 in our results (Figure 2.2) is likely an artefact of using a quadratic to model the relationship between lichen abundance and time since fire, rather than an actual decline in lichen biomass among these years. Although Thomas et al. (1996) described *C. mitis* as having a delay in recolonization of 21 to 30 years; we found no such delay in our data. The relatively rapid recolonization pattern we observed may be due, in part, to the *Sphagnum* substrate upon which lichens exist

in peatlands. Following wildfires, fragments of lichens within unburned portions of the peat substrate (which is typically quite damp and may survive incineration) could subsequently spread shortly after disturbance by wildfire. Additionally, the variability that surrounds lichen abundance among sites less than 20 years of age is substantial, and wildfire intensity may contribute to the pattern of lichen recolonization in peatlands.

Lichen Growth Rates

Our estimate of lichen growth rates are higher than most previous studies, suggesting that accelerated lichen growth contributes to the maximum biomass of lichens in peatlands being reached relatively quickly following wildfire. In northern taiga regions of Canada, Scotter (1965b) estimated annual *C. mitis* growth rates to be between 3.6-3.8 mm*year^{-1}, and in Finland growth was near 3.0-3.5 mm*year^{-1} (Helle et al 1984). Our estimated growth rate of 4.80 mm*year^{-1} is closer to the maximum recorded growth of 5.1 mm*year^{-1} in Sweden (Skunke 1969), suggesting that lichen growth rates are comparatively rapid in peatlands. Because lichen biomass was relatively constant after reaching maximum abundance though, additions to the overall lichen biomass at sites from annual growth is minimal, and the decay of lichen strand bases in the acidic peat substrate is likely compensatory to the new growth of lichens.

Growth rate of lichens increased with time since disturbance and was associated with greater presence of graminoids. We interpreted the results to be related to more xeric sites being favourable to accelerated growth of *C. mitis*. The graminoid

cover present was largely sheathed cotton-grass, which is found often in relatively dry sphagnum bogs (Moss 1983). Hence, the positive relationship of growth rate with graminoid cover may reflect faster lichen growth in moderately xeric bogs. Faster growth of lichens in older sites too, may reflect changes in bog characteristics that accompany time since wildfire. The regrowth of *Sphagnum* species post-wildfire could elevate the substrate for growth above the water table (i.e. accelerated paludification), creating more xeric microclimatic conditions favourable to lichen growth rates. Others have suggested lichen growth declines with greater cover or age (Karenlampi 1970; Thomas and Kiliaan 1998), so we conclude faster lichen growth in older sites reflects changes in site characteristics over time, rather than changes in growth due to density-dependent growth.

Implications for Woodland Caribou

There exists substantial variability in estimates of lichen biomass available for caribou among both well-drained pine and spruce stands and low-lying spruce stands in the literature; mean estimates of lichen biomass range from 420 $kg*ha^{-1}$ (Bergerud 1971) to 6276 $kg*ha^{-1}$ (Courtright 1959). Our results are the first to document lichen biomass in peatlands, and suggest that peatlands contain mean lichen abundances that are among the lowest recorded in areas caribou inhabit. Additionally, our data are generally consistent with the lower estimates of lichen biomass from spruce and pine sites randomly selected than those of caribou feeding sites. For example, lichen biomass was near 870 $kg*ha^{-1}$ (Scotter 1964, 1965b) in sites randomly selected in

Manitoba and Saskatchewan, and near 990 $kg*ha^{-1}$ in west central Alberta (Edmonds and Bloomfield 1984), but at caribou feeding sites lichen biomass was near 5487 $kg*ha^{-1}$ in Saskatchewan (Miller 1976), 4277 $kg*ha^{-1}$ in Manitoba (Miller 1976), and 4017 $kg*ha^{-1}$ in Alberta (Edmonds and Bloomfield 1984). Because resource selection occurs at multiple scales (Johnson 1980), the selection of specific foraging sites by caribou may favour sites with higher cover and biomass of lichens (Bradshaw et al. 1995; Johnson et al. 2001). The low availability of lichens in random sites that we observed likely reinforces selective behaviour by caribou for patches abundant in lichen cover.

The influence of wildfire on caribou forage resources in peatlands appears less pronounced than in other systems. Wildfires cause an immediate loss of lichens for woodland caribou, but regrowth of lichens is nearly twice as fast as regrowth in non-peatlands (Thomas and Kiliaan 1998), and caribou in Alberta already exist in an environment of low lichen availability. The variability of lichen biomass in young sites (i.e., <10 years since disturbance) suggests that wildfire intensity is variable in peatlands, so unburned island remnants within fires may also provide adequate lichen resources for caribou. Caribou in Alberta also range over very large areas annually (Bradshaw et al. 1995; Stuart-smith et al. 1997; Schneider et al. 2000), and may be able to meet winter lichen requirements in unburned portions of their range.

Just as very fine scale characteristics of peatlands are dynamic due to the constant growth and accumulation of peat, so too are the nature and patterns of peatlands at a landscape scale. Peat bogs and fens are often underlain by discontinuous

permafrost, especially in northern areas of Alberta (Vitt et al.1988), and trends of climate warming have resulted in degradation and melting of boreal permafrost over the past 100 years (Vitt et al. 1994, Halsey et al. 1995). Moreover, permafrost melting has lead to a change of forested ombotrophic bogs to non-forested poor fens (Halsey et al. 1995). The selection of ombotrophic bogs by caribou is strong among forested sites, but not evident among more open fens (Bradshaw et al. 1995, Anderson 1999), indicating that climate-induced conversion of forested bogs to more open fens could further reduce the availability of preferential caribou habitat and their forage resources.

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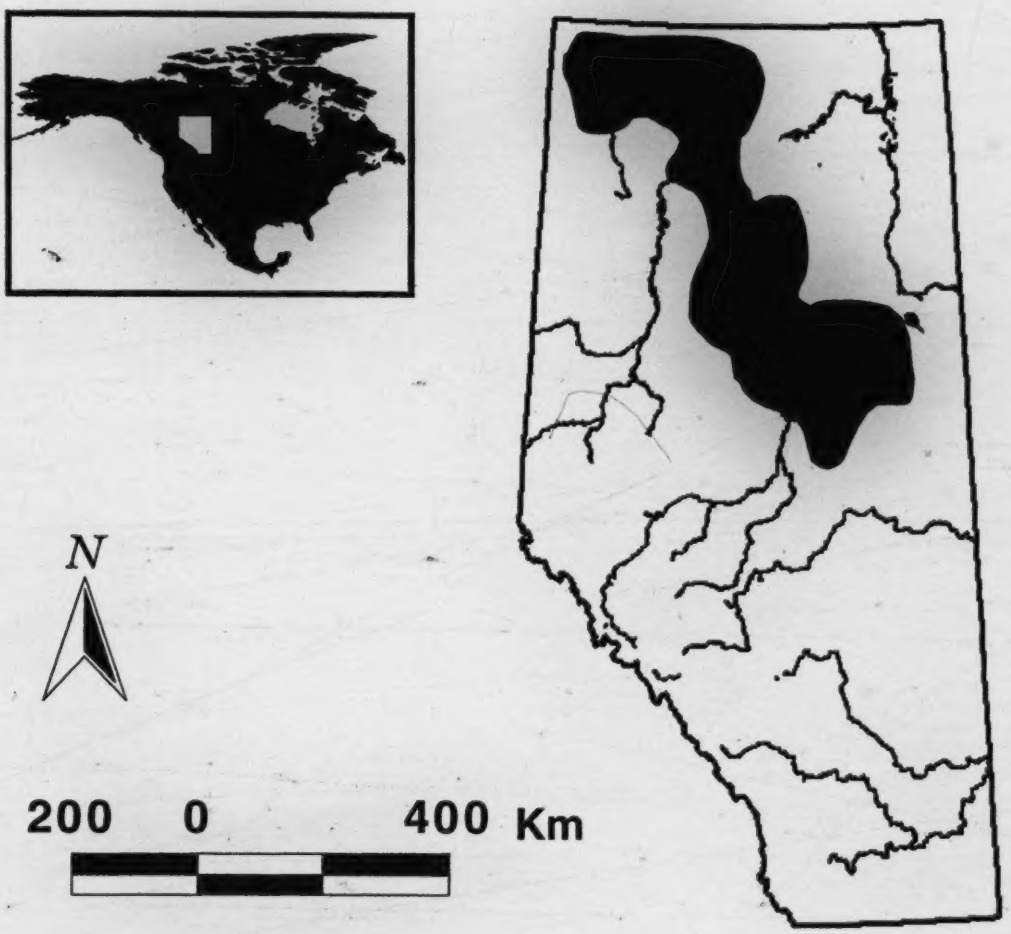

Figure 2.1 Study region within Alberta. Major rivers are shown in black, and inset shows the provincial location within North America.

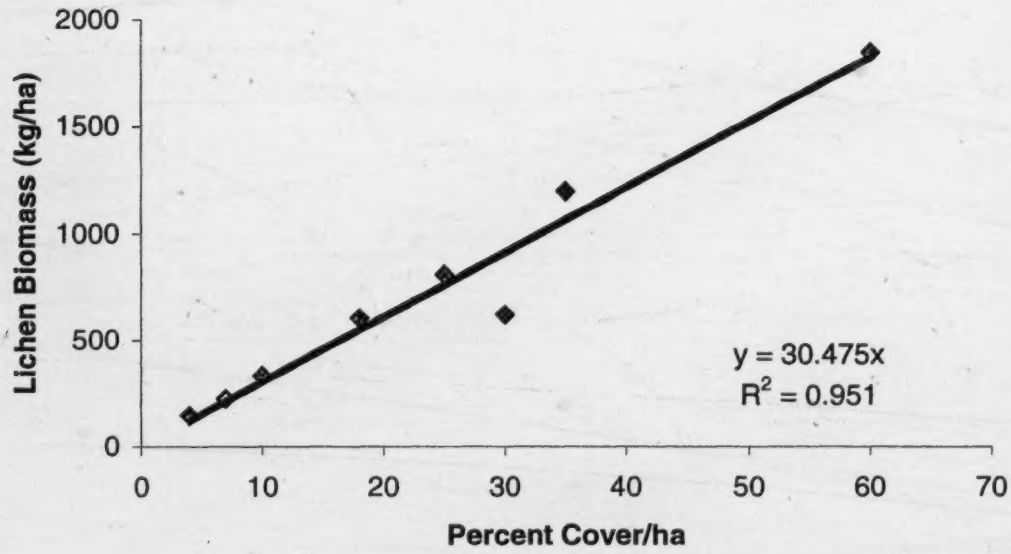

Figure 2.2 Lichen biomass-percent cover relationship for mature sites, extrapolated to a per hectare basis.

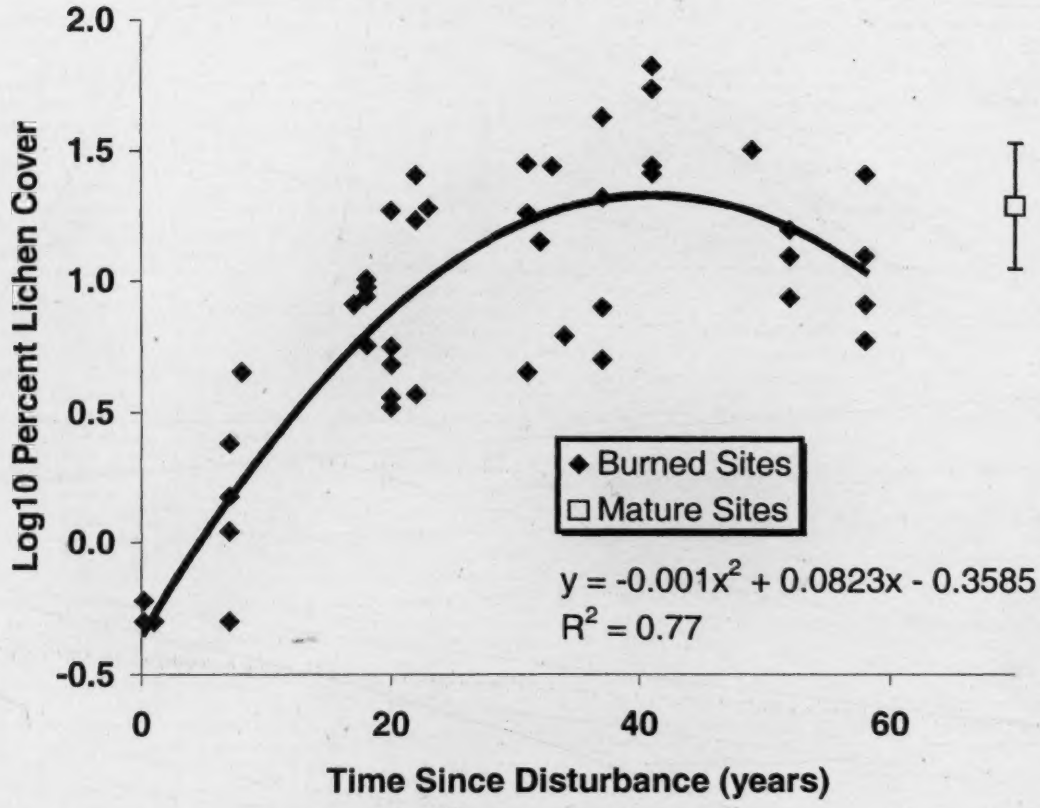

Figure 2.3 Log transformed lichen cover over time in sites of known age and mature sites (>70 years old). Error bars on mean cover estimate for mature sites are standard deviations.

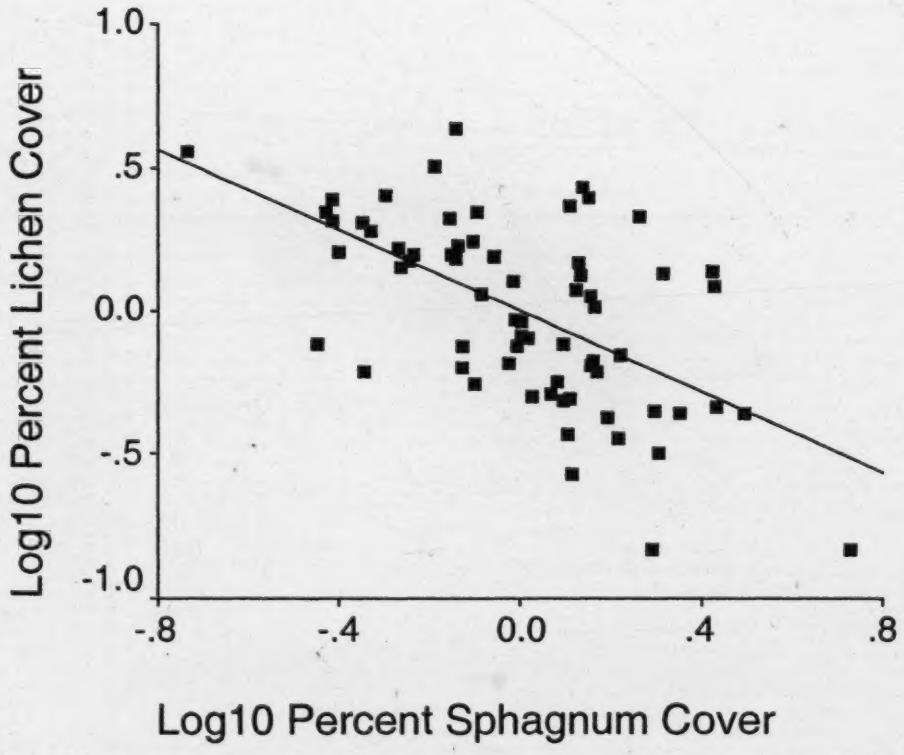

Figure 2.4 Partial plot of the relationship between lichen cover and sphagnum cover (both log transformed), independent of time since disturbance. The plot shows the residuals of the lichen cover-time since disturbance regression plotted against the residuals of the *Sphagnum* cover-time since disturbance regression.

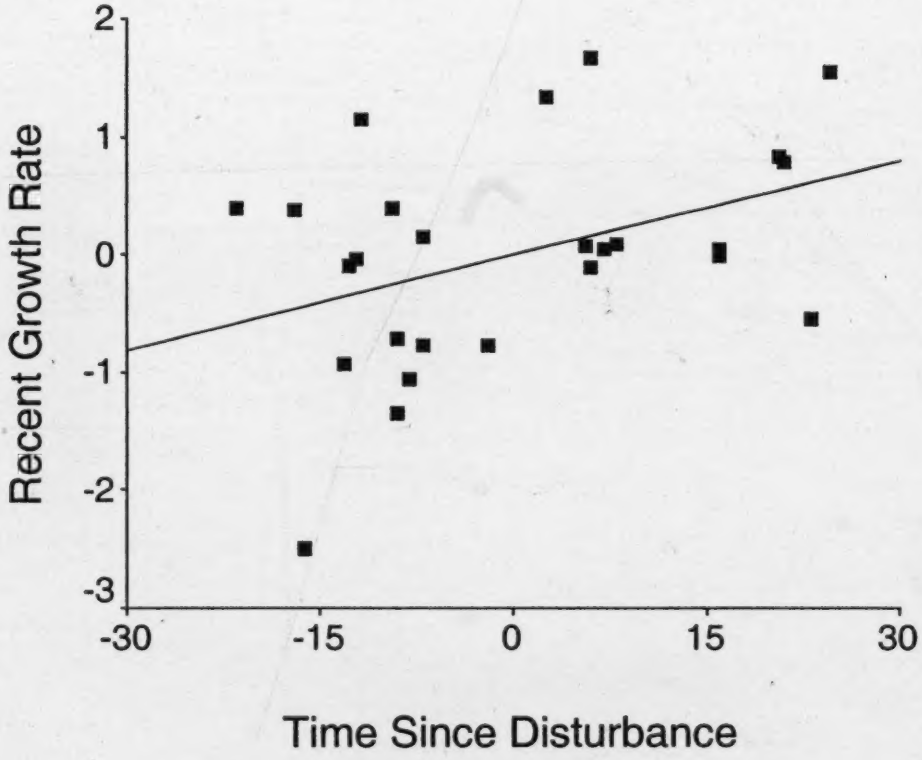

Figure 2.5 Partial plot of the relationship between recent growth rate of *C. mitis* and time since disturbance, independent of graminoid cover. The plot shows the residuals of the lichen growth rate-graminoid cover regression plotted against the residuals of the time since disturbance-graminoid cover regression

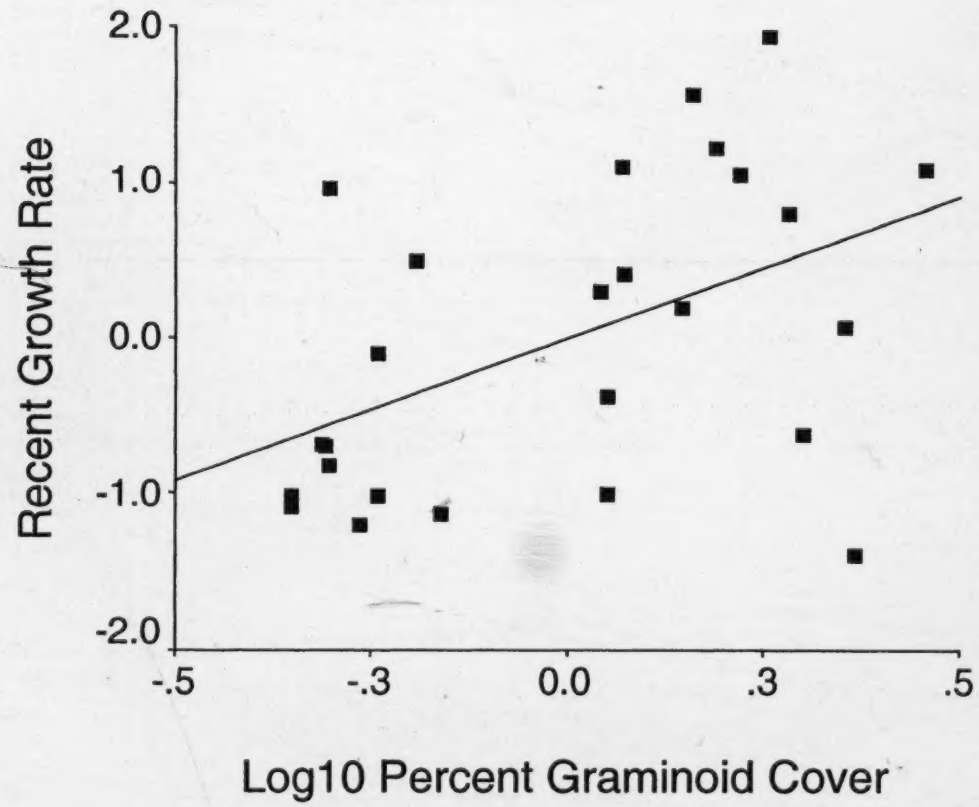

Figure 2.6 Partial plot of the relationship between recent growth rate of *C. mitis* and percent graminoid cover (log transformed), independent of time since disturbance. The plot shows the residuals of the recent growth rate-time since disturbance regression plotted against the residuals of the graminoid cover-time since disturbance regression.

CHAPTER 3. INFLUENCES OF WILDFIRE ON ANNUAL RANGE SIZE AND FIDELITY OF WOODLAND CARIBOU

Introduction

The distribution of ungulates likely reflects tradeoffs associated with energetics, availability of forage resources, predation risk, and the patchiness of resources (Fryxell et al. 1988, Kie et al. 2002). In boreal forests, wildfire events can quickly change forage availability for ungulates, altering the status of existing tradeoffs or the availability of resources. Woodland caribou (*Rangifer tarandus caribou*) are known to extensively use lichens as winter forage (Scotter 1967; Boertje 1984; Thomas et al. 1996), and the occurrence of large wildfires may result in areas within their range that offer little lichen resources. Indeed, changes in forage availability have been accompanied by changes in range use among caribou in far northern regions (Ferguson and Messier 2000).

In boreal forests though, wildfires are the dominant disturbance force (Johnson 1992), and may reduce the lichen availability for caribou for up to 40 years in northern Alberta (Dunford 2003). While smaller fires are far more common, over 97% of the area burned annually results from large wildfires (i.e., those fires >200 ha; Stocks 1991, Johnson et al. 1998). Because caribou in northern Alberta occupy very large annual ranges (Bradshaw et al. 1995, Stuart-Smith et al. 1997, Schneider et al. 2000), they are prone to having portions of their range burned by such large wildfires.

Woodland caribou concentrate on lichens to meet their forage requirements in winter (Scotter 1967; Boertje 1984; Thomas et al. 1996), so a loss in lichen availability

may result in reduced range quality (Klein1982). While lichens contain little protein (Scotter 1965a; Pulliainen 1971), they contain a reliable source of digestible energy (Cameron 1972, Pearson et al. 1975), which enables caribou to meet energy requirements during winter. Caribou are known to be selective for sites with higher lichen availability (Johnson et al. 2001), so the patchiness of lichen resources may contribute to the large annual ranges of caribou.

In boreal regions of northern Alberta, caribou are non-migratory, wide-ranging, and select peatland complexes (treed bogs and fens; Bradshaw et al. 1995; Anderson 1999; Schneider et al. 2000). The preference for peatlands by boreal caribou in Alberta has been hypothesized as a strategy to reduce predation risk; although not rigorously tested, selection of low-lying peatlands by caribou may reduce exposure to wolves (*Canis lupus*) and the alternate ungulate prey of wolves that are found primarily in well-drained uplands and river valleys (moose; *Alces alces,* white-tailed deer; *Odocoileus virginianus*, mule deer; *O. hemionus*, James 1999; Fuller and Keith 1980; Hauge and Keith 1981; Bradshaw et al. 1995; Stuart-Smith et al. 1997; Anderson 1999; Schneider et al. 2000). Additionally, the fidelity of caribou to sites and ranges can be viewed as predator avoidance behaviour (Rettie and Messier 2000, Schaefer et al. 2000).

Here we examined whether a substantial change in winter forage availability resulting from wildfires altered the fidelity and distribution of caribou. We tested whether (1) caribou had differing range fidelity in years with large wildfires, (2) the extent of both wildfire and peatland abundance in ranges influenced range reuse, and

(3) the size of annual caribou ranges differed before and after large wildfire events. Because wildfires result in a loss of lichen availability for caribou, we expected caribou distribution to reflect behaviour that compensated for such losses, and predicted that following wildfires, caribou would shift ranges to incorporate new areas in their annual range, so range fidelity should be reduced following wildfires. We reasoned that the proportion of annual range burned would be negatively correlated with range reuse, and that the size of annual ranges may increase following fires, because caribou may range over a larger area to meet lichen requirements. Because use of peatlands may be an important means of reducing predation risk, we predicted greater peatland abundance within caribou ranges should be positively correlated with range fidelity.

Study Areas

Our study areas were three regions of northern Alberta inhabited by woodland caribou and where large wildfires have recently occurred (Figure 3.1). Herein, we refer to these regions as the Caribou Mountains (CM), East Side of the Athabasca River (ESAR), and Red Earth (RE). The CM region is largely within the boreal sub-arctic and wetland mixedwood natural subregions, while the RE and ESAR regions were part of the central mixedwood natural subregion (Alberta Environment 2002). All regions contain poorly drained peatlands in low-lying areas and well-drained uplands and river valleys. Peatlands support black spruce (*Picea mariana*) and larch (*Larix larcina*) trees, and understory vegetation that includes Labrador tea (*Ledum groenlandicum*),

sheathed cotton-grass (*Eriophorum vaginatum vaginatum*), and *Sphagnum* mosses (Vitt et al. 1996). Uplands are dominated by stands of white spruce (*Picea glauca*), jack pine (*Pinus banksiana*), and trembling aspen (*Populus tremuloides*), and shrubs such as wild rose (*Rosa acicularis*) and alder (*Alnus crispa*). Topographic variation is minimal in all regions, although elevation is greater in the CM region.

In 1995, wildfire burned approximately 170,000 ha in CM, 140,000 ha in ESAR, and just over 180,000 ha of the RE region in 1998. In ESAR and RE, little else of the region had been burned by wildfires other than in the fires of 1998 and 1995, respectively. In CM, previous wildfires from the early 1980's had burned an area approximately equal in size to the 1995 fires.

Methods

Data Collection

Caribou location data used to estimate distribution was collected as part of a larger research program designed to identify caribou population trends, habitat associations, and measure influences of disturbance, including wildfires, on caribou (Bradshaw et al. 1995; Stuart-Smith et al. 1997; James and Stuart-Smith 2000; Schneider et al. 2000; Dyer et al. 2001; Dyer et al. 2002; McLoughlin et al. 2003). Caribou were equipped with very high frequency (VHF) radio-collars (Lotek Engineering Inc., Newmarket, Ontario, Canada), and relocated bi-weekly in CM and RE, and monthly in ESAR using fixed-wing aircraft, and locations recorded using

GPS receivers. Because of an emphasis to assess caribou population trends, almost all collared caribou were female (McLoughlin et al. 2003).

In ESAR, location data was collected from 1992-2001, yielding 3 years of pre-fire data (n = 72 annual ranges from 31 individuals) and 6 years post-fire data (n = 134 from 44 individuals). In CM and RE, caribou telemetry data was collected from 1995-2001. Only 6 months of pre-fire telemetry data was available in CM (n = 15 six-month ranges from 15 individuals), but 6 years of post-fire data (n = 91annual ranges from 28 individuals). In the RE region there were 4 years of pre-fire data (n = 71 annual ranges from 31 individuals), and 3 years of post-fire data (n = 62 annual ranges from 33 individuals).

Home range calculation

Annual caribou home ranges were calculated from caribou telemetry data using the Animal Movement extension (Hooge and Eichenlaub 1997) in ArcView 3.1 (Environmental Systems Research Institute Inc. 1998). We used a 95% fixed-kernel estimate of annual ranges, using the least squares cross-validated method (Silverman 1986; Worton 1989, Seaman and Powell 1996; Seaman et al. 1999). Annual home ranges were based on relocations from November 1st to October 31st of the following year, in order to approximate a biologically meaningful year spanning the early winter to the end of rutting (Bergerud 1975). The number of locations used to estimate annual home ranges was 24.6 in CM, 24.1 in RE, and 12.6 in ESAR; prior analysis indicated that annual home range size asymptoted within 20 locations (unpublished data).

Statistical Analysis

We measured home range shift by the proportion of the previous year's home that was included in the following year's home range. We used a repeated measures analysis of variance (ANOVA) to test if year was a significant factor in the overlap between consecutive annual caribou home ranges, and a multiple linear regression to compare the percent home range overlap following fire with the proportion of the annual range burned and proportion annual range composed of peatland. We present pooled data from the ANOVA because no differences existed in range overlap among regions. Both the percent home range overlap and peatland composition were log transformed to meet assumptions of normality in the regression.

To assess whether an occurrence of wildfire influenced home range size, we used a paired *t*-test to compare the change in mean annual home range size of caribou before and after a wildfire event. For paired comparisons, we used only those caribou that had a portion of their annual range burned, and compared their annual range size before and after wildfires. Because only 6 months of pre-fire data was available in the CM region, we compared the size of home ranges between the 6-month periods before and after wildfire in that region.

Results

While caribou showed variability in annual range fidelity, year was not a significant factor in the ANOVA ($F = 0.26_{8,24}$, $p = 0.97$; pooled data), indicating that

range reuse following a year with wildfires did not differ from years in which no fires occurred (Figure 3.2). Among regions, average reuse of individual ranges was very similar. In CM, the mean overlap of consecutive annual ranges was 58.5% (SE = 1.9, Range 10.3% to 96.7%, n = 106), in ESAR 58.6% (SE = 1.8, Range 0.6% to 100%, n = 206), and in RE 61.3% (SE = 2.0, Range 3.7% to 100%, n = 133).

The proportion of range burned showed no relationship with annual range reuse (P = 0.48; Figure 3.3), while the proportion of peatland within annual home ranges showed a marginally non-significant relationship with range reuse (P = 0.06; Figure 3.4). Residual plots (Figures 3.3 and 3.4) depict the relationship between the dependent variable and the independent variables controlling for other variables in the multiple regression. The direction of the relationship between peatland composition of annual home ranges and range reuse was slightly negative; suggesting that greater peatland composition within ranges could result in less reuse in the following year. However, the regression of percent home range reuse on percent annual range burned and percent peatland composition gave a poor model fit (adjusted r^2 = 0.12), indicating that these two variables alone did not explain a substantial proportion of the variation in annual range fidelity we observed.

Substantial variation in home range size was evident among regions; caribou in the CM region had home ranges greater than twice the size of caribou in ESAR or RE. However, differences in home range size before and after fire events were non significant in each range (Figure 3.5; CM, t = 0.191, df = 14, P = 0.85; ESAR, t = -0.628, df = 24, P = 0.54; RE, t = 0.374, df = 17, P = 0.71).

Discussion

Range Fidelity

We expected caribou to shift annual ranges in response to a loss of winter forage, but the degree of fidelity obsevered in years with wildfires did not differ from years without fires, even though fires burned up to 40% of caribou ranges. Our estimates of range fidelity are similar to the results of previous studies (e.g., Stuart-Smith et al. 1997, Rettie and Messier 2001); however, this study is the first attempt to document factors that may influence fidelity for caribou in Alberta. While we were not able to explain most of the variation observed in range fidelity, our results suggest that the occurence of wildfire has little relationship with the reuse of annual ranges. Because lichen loss increases with the extent to which ranges burned, we expected range fidelity to decrease with the proportion of range burned, but this prediction was not supported. Caribou may reuse ranges as a means of minimizing predation risk (Schaefer et al. 2000), even if wildfires reduce lichen availability. Alternatively, better productivity of summer forages following wildfire (Schaefer and Pruitt 1991) may negate the loss of lichens from wildfire.

The negative relationship observed between range fidelity and peatland abundance was unexpected (albeit non-significant), and contrary to expected behaviour that might reduce predation risk for caribou, because peatlands are hypothesized to offer partial refugia from predation for caribou (James 1999). The strength of the relationship we observed was relatively weak however, so peatland

abundance within ranges may also be independent of fidelity. If the negative relationship we found is true, we speculate that it could be a reflection of the risks that may be associated with movement through non-peatlands. That is, caribou with little peatland abundance within their range may elect to exist soley within smaller peatland patches, rather than risk travel though predator-dominated uplands to expand their range. Conversly, caribou that have ranges containing predominately peatland habitats may be less limited by movement risks, and are free to travel and exploit other areas on an annual basis, resulting in decreased fidelity.

Range Size

We detected no difference in range sizes before and after the occurance of wildfire, hence we suggest that caribou forage needs may already be met within the relatively large areas used, even with the reduction in lichen biomass resulting from widfire. Because caribou are more widely distributed in winter (Fuller and Keith 1981, Stuart-Smith et al. 1997, Rettie and Messier 2001), our annual range estimates probably better reflect caribou winter distribution than those of the summer. Lichen availabilty in peatlands habitats is relatively low (Dunford 2003), so the wide distribution of caribou in winter may indicate selective behaviour already in place that meets forage requirements. Further reduction in lichen availability may be inconsequental, because caribou may select for unburned lichen patches within burned areas, or use alternate areas of their range to meet forage needs. The relatively large annual home ranges we observed are consistent with other studies that relate ungulate

home range size with food availability, whereby home range size increases with decreasing food availability (Clutton-Brock et al. 1982, Tufto et al. 1996).

Our measures of annual range size for caribou in the ESAR and RE regions are comparable with those of other authors (Bradshaw et al. 1995, Stuart-Smith et al. 1997,Schneider et al. 2000). In ESAR, we may have underestimated annual range size because our telemetry location sample size for individuals was poor, although our estimates of home range size in this region do not appear substansially different than those of previous works (e.g., Bradshaw et al. 1995, Stuart-Smith et al. 1997; Schneider et al. 2000). This study is the first to report home range sizes for caribou in the far northern areas of Alberta (CM region), and our results clearly imply that caribou in this region are more widley distributed than caribou in more southern regions. Because CM region has experienced previous wildfires in the early 1980's, the large annual ranges of caribou in this region may reflect selection for patches of high lichen availability (Bradshaw et al. 1995, Johnson et al. 2001) that are likely more sparse as a result of wildfires.

Conclusion

Our observations were inconsistent with our predictions; changes in forage availability resulting from wildfire did not alter caribou distribution or range fidelity. Caribou showed no difference in range fidelity in years with fires from those without disturbances, and the extent to which ranges burned was unrelated to fidelity, nor did annual range size change following wildfires. These data suggest that losses of winter

forage from wildfires (at the scale we observed) are independent of caribou distribution. Caribou show selection for peatland complexes (Bradshaw et al. 1995, Stuart-Smith et al. 1997, Anderson 1999, Schneider et al. 2000), so the role of peatland selection may be important in determining distribution of caribou at this scale. It is hypothesized that caribou select peatland complexes because they allow for spatial separation with the alternate prey of wolves (James 1999). Hence, annual ranges by caribou may reflect home ranges that closely mirror the shape and size of peatland complexes (Stuart-Smith et al. 1997). If so, the very large annual ranges in the CM region may be explained well by the underlying peatland patterns in the region. Future research aimed at determining the extent to which peatland patterns influence caribou distribution would be valuable to wildlife managers. Furthermore, the consistent range fidelity we observed might result from behaviour that reduces predation risk for caribou at this scale, reinforcing the notion that fidelity reflects behaviour consistent with predator avoidance (Schaefer et al. 2000).

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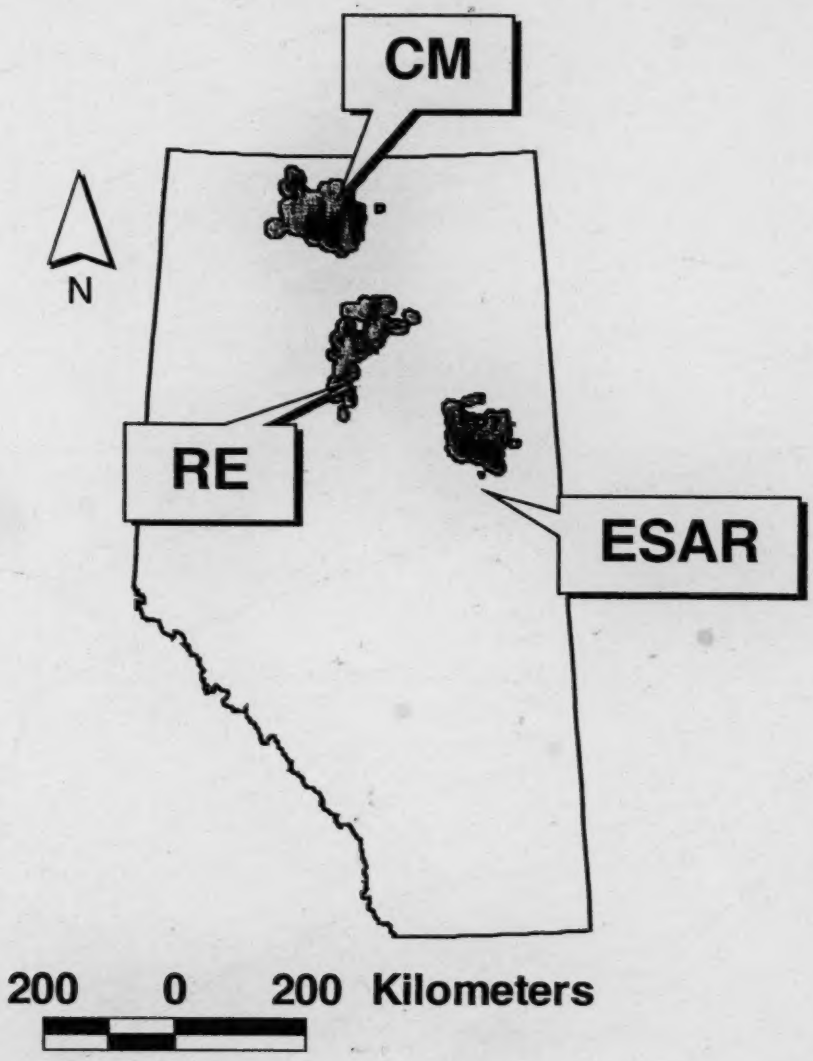

Figure 3.1 Three study areas within the province of Alberta, Canada. The acronyms CM, RE, and ESAR refers to the regions Caribou Mountains, Red Earth, and East Side Athabasca River, respectively.

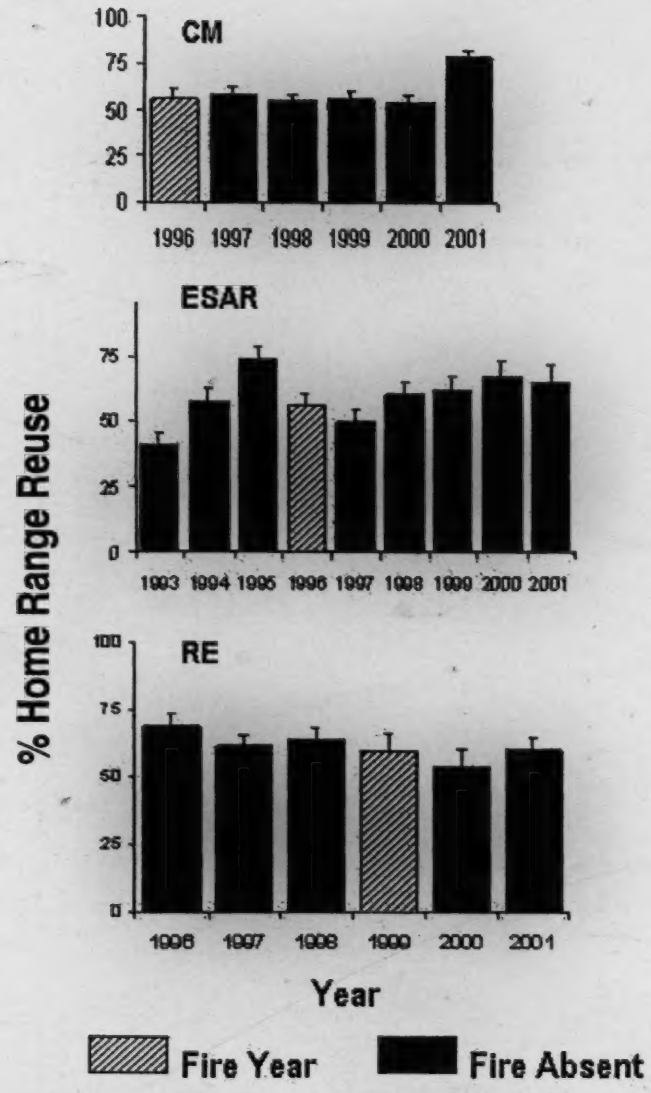

Figure 3.2 Mean annual home range reuse in the CM, ESAR, and RE regions. Error bars represent standard errors measures, and "Fire Year" refers to the year immediately following a large wildfire event. "Fire Absent" refers to years without wildfires of any appreciable size.

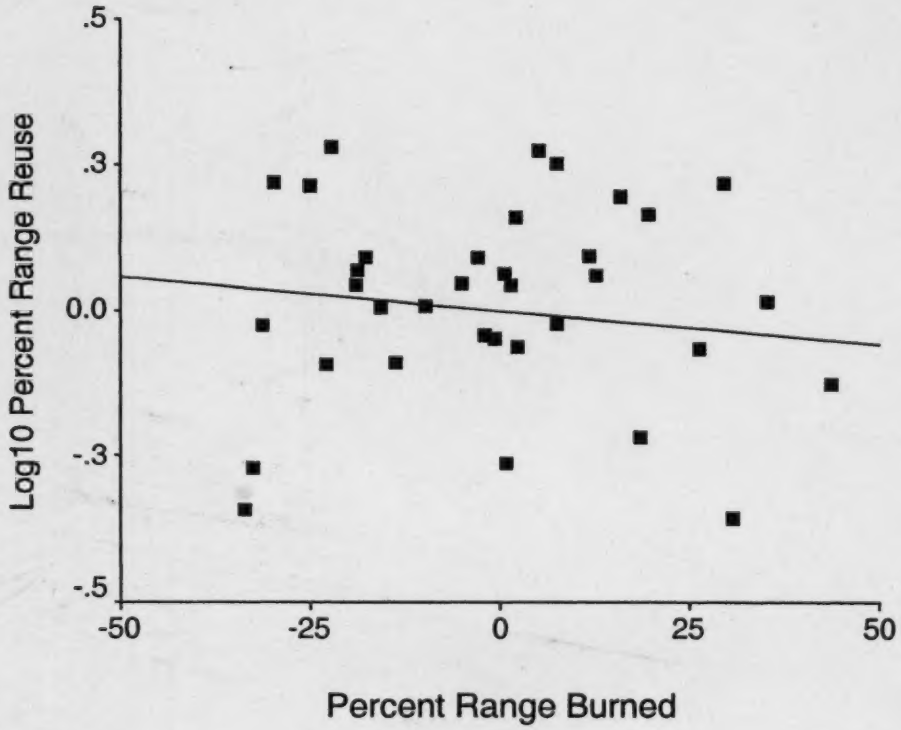

Figure 3.3 Partial plot the relationship between range reuse and the percent ranges were burned, independent of peatland composition within ranges. The plot shows the residuals of the percent range reuse-peatland composition regression (both log transformed) plotted against the residuals of the percent range burned-peatland composition within ranges (log transformed) regression.

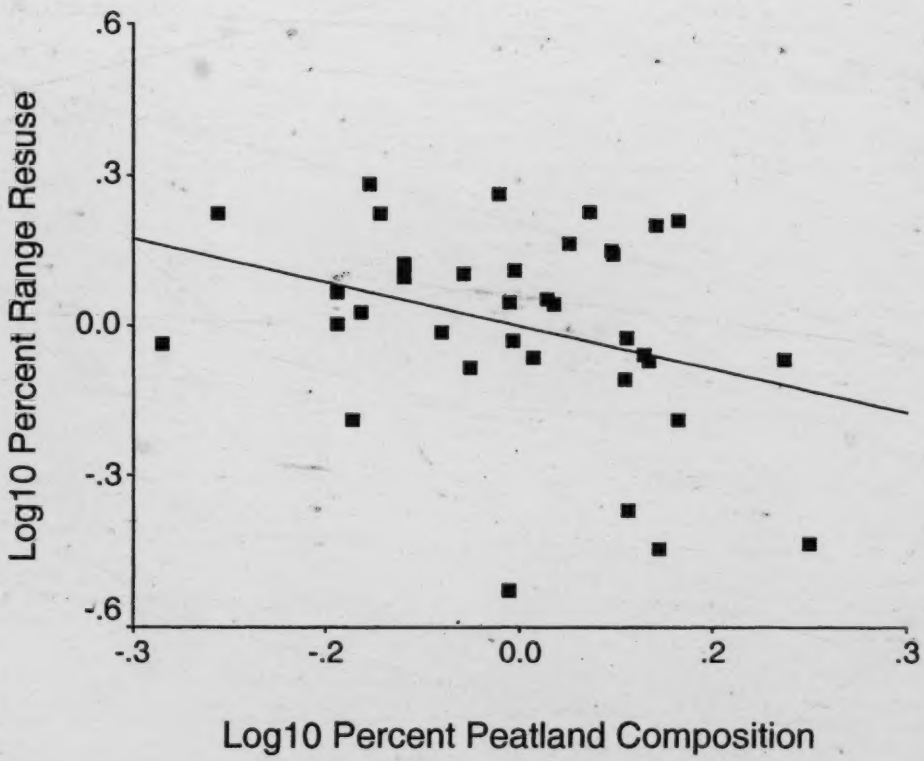

Figure 3.4 Partial plot the relationship between range reuse and peatland composition within ranges (both log transformed), independent of the amount ranges were burned. The plot shows the residuals of the percent range reuse-percent range burned regression (both log transformed) plotted against the residuals of the log transformed percent peatland composition-percent home range burned regression.

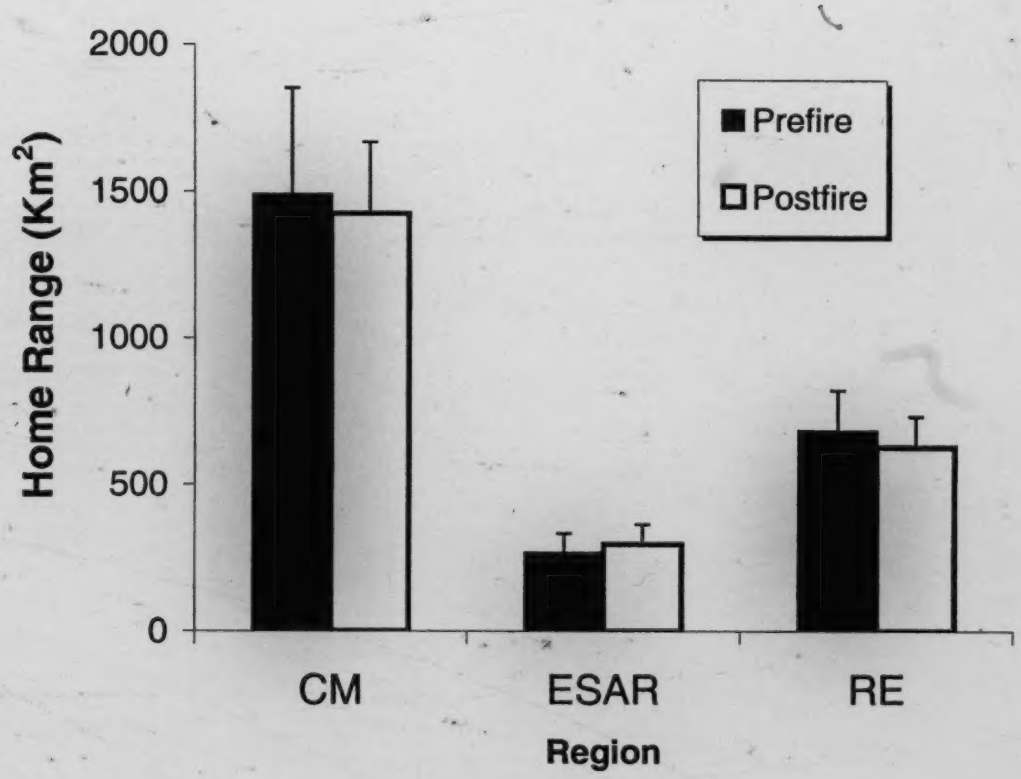

Figure 3.5 Paired home range size comparison before and after fire events among the Caribou Mountains (CM), East Side Athabasca River (ESAR), and Red Earth (RE) regions. Error bars represent one standard error.

CHAPTER 4. WILDFIRE INFLUENCES ON HABITAT USE AND SELECTION BY WOODLAND CARIBOU

Introduction

Ecological tradeoffs resulting from relative habitat use may be apparent for animals with differing availabilities of habitat types (Mysterud and Ims 1998). For wide-ranging animals such as ungulates, disturbance processes can quickly alter the availability of habitat types, but habitat selection has not yet been shown to be a consistent function of habitat availability for such animals. In boreal forests, wildfires are the principal disturbance force, resetting plant community succession, and maintaining the diversity and heterogeneity of habitat types (Johnson 1992). Wildfires also can quickly change the availability of animal forages associated with the age of forest stands. For woodland caribou in Alberta (*Rangifer tarandus caribou*), wildfires reduce the availability of lichen in late succession habitats (Dunford 2003).

In northern Alberta, woodland caribou have large annual ranges that encompass peatland complexes (i.e., >500 km^2; Bradshaw et al. 1995; Stuart-smith et al. 1997; Schneider et al. 2000). The preference for peatlands by caribou in Alberta has been hypothesized as a spatial separation strategy: Wolves (*Canis lupus*) are the foremost predator of caribou, but generally occur outside of peatland habitats. Other ungulate prey of wolves, such as moose (*Alces alces*) and deer (white-tailed deer; *Odocoileus virginianus*, mule deer; *Odocoileus hemionus*) are more prominent in wolf diets, and are found in uplands and well-drained river valleys (James 1999; Fuller and Keith 1980; Hauge and Keith 1981; Bradshaw et al. 1995; Stuart-Smith et al. 1997;

Anderson 1999; Schneider et al. 2000). Preference for peatlands suggests that at the scale of the population caribou select habitats that reduce predation risk, because of the limiting potential of predation (Rettie and Messier 2000). Selection based on seral stage or lichen availability is not apparent at these larger spatial scales (unpublished data).

Lichens are the primary source of winter forage for woodland caribou (Scotter 1967; Boertje 1984; Thomas et al. 1996), but wildfires can reduce the availability of lichens for up to 40 years in northern Alberta (Dunford 2003). Hence, caribou are known to favour foraging sites of older stands that are rich in lichens (Edmonds and Bloomfield 1984; Bradshaw et al. 1995; Johnson et al. 2001). While lichens contain little protein (Scotter 1965a; Pulliainen 1971), they contain a reliable source of energy for caribou in winter (Cameron 1972, Pearson et al. 1975). Other forages more rich in nutrients are more readily available in summer and in younger habitats. Lichens tend to compose a lesser component of caribou diets in the snow-free season (Boertje 1984; Thomas et al. 1996) and it is assumed that selection for older habitats is less prevalent in summer (Bradshaw et al. 1995).

Recent work that examined caribou distribution in relation to wildfires found that home range size and fidelity was independent of the occurrence of fires (Dunford 2003). Presumably, caribou do not shift ranges in response to fires, because they value the large peatland complexes they inhabit for their predator avoidance benefits. Because wildfires reduce lichen availability in peatlands though (Dunford 2003), we questioned whether caribou had strategies (other than distributional changes) to cope

with wildfire occurrence. Among migratory caribou, caribou have been shown to move through burned regions, but spend little time foraging in such regions (Thomas et al. 1998). If non-migratory caribou in Alberta are bound to peatland regions, they may be forced to use burned areas when these areas form a large component of their home range.

Here we document caribou habitat use and selection in four regions of Alberta that had varying amounts of recent wildfire occurrence. Our intentions were to determine to what extent caribou use and select seral stages within their annual ranges, and if selection varies with availability. Because caribou depend on lichens more so in winter, we evaluate caribou use and selection of habitats among the winter and summer seasons. Our study compares habitat use and selection in three seral stages (young, mature, and old) among well-defined peatland and uplands habitat types.

Study Areas

Our study areas include four regions occupied by boreal woodland caribou within northern Alberta, Canada (Figure 4.1). Herein, we refer to these regions as Caribou Mountains (CM), East Side of the Athabasca River (ESAR), West Side of the Athabasca River (WSAR), and Red Earth (RE). The WSAR and ESAR regions have relatively low amounts of recently burned habitats, while the RE and CM regions have a greater abundance of recent wildfires.

These regions are naturally fragmented into low-lying peatlands and well-drained uplands. Peatlands range in size from 10 to >100,000 ha, and support black

spruce (*Picea mariana*) and larch (*Larix larcina*) trees. Understory vegetation in peatlands is typically Labrador tea (*Ledum groenlandicum*) and *Sphagnum* mosses (Vitt et al. 1998). Uplands support stands of white spruce (*Picea glauca*), jack pine (*Pinus banksiana*), and trembling aspen (*Populus tremuloides*), and shrubs such as wild rose (*Rosa acicularis*) and alder (*Alnus crispa*). Topographic variation is minimal in all regions, although elevation is greater in the Caribou Mountains region.

Methods

Data Collection

Location data for caribou was collected as part of a larger research program designed to identify population trends, caribou-habitat associations, and integrate measures of disturbance into cumulative effects models (Bradshaw et al. 1995; Stuart-Smith et al. 1997; James and Stuart-Smith 2000; Schneider et al. 2000; Dyer et al. 2001; Dyer et al. 2002; McLoughlin et al. 2003). Almost all caribou represented in this study were female. Caribou were equipped with very high frequency (VHF) radio-collars (Lotek Engineering Inc., Newmarket, Ontario, Canada). From 1991-2001, caribou were located approximately biweekly using fixed-wing aircraft, and caribou locations were recorded using GPS receivers.

The Peatland Inventory of Alberta (Vitt et al. 1996) was used to classify the landscape into patches of peatland or non-peatland. Here we refer to non-peatlands as uplands, because they were commonly dominated by species that occurred on well-drained soils. The Peatland Inventory is based on homogeneous polygons of

vegetation types identified from aerial photography. Each polygon within the inventory is categorized according to a hierarchical classification system that reflects the percentage composition of different wetland types. Based on caribou habitat selection studies (Schneider et al. 2000; Bradshaw et al. 1995), we classified polygons into peatlands if 50% of a given polygon was composed of fen or bog classes.

Caribou home ranges were calculated from location data using the Animal Movement extension (Hooge and Eichenlaub 1997) in ArcView 3.1 (Environmental Systems Research Institute Inc. 1998). We used a 95% fixed kernel estimate of home ranges, using the least squares cross-validated method (Silverman 1986; Worton 1989, Seaman and Powell 1996; Seaman et al. 1999).

We classified stand age from wildfire history maps (Government of Alberta 2002), and established three seral stages: 0 to 10 years (young), 11 to 50 years (mature), and 51+ years (old). Our goal was to delineate habitats with very low lichen abundance (young) and high lichen abundance (old), and to ensure habitat availability in each class that composed annual ranges was > 5% (hence only one intermediate class; mature). Changes in seral stages due to forest harvest were not considered in the habitat classification because most areas of caribou range are peatlands, which contain little merchantable timber. Annual home ranges were generated based on telemetry data from the period 1 November - 31 October, a 12-month period spanning early winter to the end of rutting (Bergerud 1975). Seasonal home ranges were calculated for 1 May - 31 October (summer, approximating the snow-free period of calving to the end of rutting), and 1 November - 31 April (winter; Bergerud 1975).

Habitat use and selection

We estimated habitat use within the home range by the proportion of caribou telemetry locations within each habitat class. To estimate caribou selection of habitats within annual ranges, we calculated a resource selection function (Manly et al. 1993), using the proportional composition of individual home ranges as our measure of habitat availability (Table 4.1). We only included caribou for which we had at least 35 relocations from which to estimate a home range (n = 178). Caribou use was considered 'avoidance' if habitat types were used proportionally less than availability, 'preference' if habitat types were used proportionally more than availability, and 'random' if habitat use occurred in proportion to availability (Johnson 1980).

Habitat composition was calculated using the Spatial Analyst extension for ArcView (Environmental Systems Research Institute Inc. 1998). Selection ratios for each habitat type among caribou were expressed as:

$$w_i = \frac{proportion\ \ used_i}{proportion\ \ available_i}$$

and then standardized using the following equation:

$$b_i = \frac{w_i}{\sum_{i=1}^{H} w_i}$$

where i ranged from 1 to H, and H represented the number of habitat types. The set of beta coefficients (b_i values) for individual caribou represented the unit of replication for expressing mean selection indices of the habitat types. Beta coefficients presented in the resource selection function (the set of b_i's) can be interpreted as the probability that the associated habitat type would be chosen in a selection event, assuming proportionately equal availabilities of all habitat types (Manly et al. 1993).

We tested for significant differences in selection between uplands and peatlands using Bonferroni-corrected multiple contrasts, and assigned significant preference or avoidance to habitat use if 95% confidence intervals of the beta coefficients did not overlap the expected beta coefficients of a random use model. Thus, we do not present P - values for inferences of selection and avoidance, but because we used 95% confidence intervals to measure selection, one can assume significance at an alpha of 0.05 if habitat use is termed 'preferred' or 'avoided'.

Habitat selection by season

We also used a multivariate analysis of variance (MANOVA) to test whether the habitat selection pattern of caribou differed by season. The overall caribou selection pattern was described by H beta coefficients ($H = 6$ in our case), but because these were correlated by definition (they summed to 1), H - 1 synthetic variables were used to describe the habitat selection pattern (i.e., dependent variables) in the MANOVA (Arthur et al. 1996). Synthetic variables were calculated as the selection coefficient of habitat type i minus the selection coefficient of habitat type i - 1. We

then tested the factor 'winter or summer' to determine whether patterns of habitat selection within caribou home ranges differed during these two seasons.

Results

Within the four regions studied, caribou were most often located in the old peatland category (Table 4.2). Habitat use appeared similar among the WSAR and ESAR regions, and among the RE and CM regions. In WSAR and ESAR, caribou were found less than 12% of the time in all young and mature classes combined, and old peatlands were used more than old uplands (nearly 3 times more in WSAR and nearly 2 times more in ESAR). In RE and CM, caribou used young and mature habitats twice as much as caribou in WSAR and ESAR. Use of old peatlands was similar; caribou were found in this habitat approximately ¼ of the time in both regions.

Caribou in both ESAR and WSAR showed avoidance of the young habitats, and preference for old habitats (Figures 4.2 and 4.3). Caribou in WSAR used the mature peatland habitats randomly and avoided the mature upland class, while those in ESAR avoided all mature habitats. In ESAR, significant differences in selection for peatland and uplands within seral stages were evident only in the old seral stage ($P <$ 0.00001 for both summer and winter). In WSAR, differences in selection for peatland and uplands within seral stages was also present for the old seral stage ($P < 0.00001$ for both summer and winter), but also in the mature seral stage in winter ($P = 0.0067$).

In RE, the young peatland and upland classes were avoided, and the old peatland and upland classes were preferred (Figure 4.4). Mature peatlands were used randomly, while mature uplands were avoided. No significant differences in selection between peatlands and uplands within seral stages were evident in either summer or winter.

In CM, selection tended towards more random use, as selection coefficients deviated less from the null selection model than selection in other ranges. Young and old peatlands were used randomly in summer, while mature peatlands were preferred (Figure 4.5). In winter though, peatlands were preferred in the young and old classes, but avoided in the mature class. Young and old uplands were avoided in summer, but mature uplands were preferred in summer. In winter, uplands were used randomly in all seral stages. Within seral stages in both winter and summer, no significant differences between upland and peatland selection were detected.

Seasonal selection patterns did not differ in ESAR ($F_{5, 142} = 1.18$, $P = 0.32$), WSAR ($F_{4, 152} = 1.40$, $P = 0.24$), or RE ($F_{5, 98} = 0.49$, $P = 0.78$). Caribou in the CM region however, did show significant differences in seasonal selection patterns ($F_{5, 89} = 6.10$, $p < 0.00001$). Caribou in CM show greater proportional use of young and old habitats in winter, and decreased proportional use of mature habitats in summer.

In general, we observed that caribou avoided young and mature habitats of both peatland and upland at the home range scale. In most areas, old habitats of both upland and peatland were consistently preferred. In the CM however, young habitats composed over 21% of annual ranges, and habitat use appeared more random. Caribou

tended to select peatlands over uplands, and seasonal changes in selection were not evident in most regions.

Discussion

Our data suggest that within annual ranges, caribou generally selected older seral stage habitats and seasonality of selection was absent. Lichen availability is relatively low in peatlands (Dunford 2003), so the incentive for selection of habitats with greater lichen biomass (i.e., raised xeric bogs; Bradshaw et al. 1995) may be more critical for caribou in northern Alberta than those caribou in other regions with more abundant lichens or with seasonal ranges. Large annual ranges and fidelity to ranges by caribou appears to vary little with wildfire occurrence (Dunford 2003), so the absence of seasonality in selection patterns could be a product of year-round use of specific peatland complexes by caribou. Selection of older habitats likely results from a dietary need for lichens in winter, but if the use of peatlands reduces predation risk for caribou (James 1999, Dunford et al. *In Review*), the lack of seasonal selection patterns by caribou may be evidence of predator avoidance behaviours rather than forage-based habitat choice alone.

Notwithstanding the importance of older habitats to caribou for their forage benefits, we observed increased use of younger habitats in regions with greater abundance of wildfires, and selective habitat use that varied with the availability of habitat types. An increase in availability of younger habitats coincided with a decreased avoidance of those habitat types. Two possibilities exist that may explain

this pattern: First, the value of different habitat types for caribou may be a function of the availability of such habitats, and habitat selection patterns reflect such changing habitat values. Second, habitat selection patterns vary regionally in Alberta, and the differences we observed may have resulted from the differences in caribou behaviour among regions.

Relative habitat availability may lead to trade-off situations for caribou that are reflected in different selection patterns (Mysterud and Ims 1998). With limited presence of recent wildfires, caribou may chose to avoid burned areas completely, because adequate lichen biomass more easily found in larger, contiguous patches of older habitats. When wildfires compose a larger portion of annual ranges, caribou may forage within burns for residual unburned islands, and selection measures indicate reduced avoidance of burns. Caribou in Alberta may also use burns as corridors, but spend little time foraging in such areas, similar to migratory caribou (Thomas et al. 1998). Caribou also select the interior core of peatland complexes because that selection appears to reduce predation risk (Dunford et al., *In review*); if wildfire presence is minimal, caribou may be able to avoid burns without compromising the use of interior peatland regions. If wildfires are more abundant however, caribou may not be able to avoid recent wildfires without compromising their use of core peatland complexes, and hence show decreased avoidance of burns with greater wildfire presence.

Alternatively, decreased avoidance of younger stands with their increasing availability may be attributable to the characteristics of habitats within regions. In the

CM region for example, large wildfires occurred in the early 1980's. Moose and predator densities may be relatively high in the CM if populations of moose increased over time since wildfire (Peek 1974; MacCracken and Viereck 1990). Caribou are much more widely distributed in the CM (i.e., annual ranges >2000 km^2; Dunford 2003), possibly as a means to reduce exposure to predators. The very large annual ranges of caribou may have resulted in reduced avoidance of young habitats as caribou travel throughout the region. In the RE region, peatlands tended to be more fragmented and in smaller patches, and caribou there may also travel regularly though young habitats to reach peatland patches, giving the appearance of relatively reduced avoidance of younger habitats.

Bergerud et al. (1990) proposed that woodland caribou "forage optimally in a fine-grained sense and reduce [predation] risk in a coarse-grain manner." Recent work has refined this premise to argue that caribou habitat selection is spatially and temporally hierarchical in response to limiting factors (Rettie and Messier 2000). Boreal populations of caribou are thought to be limited by predation (Seip 1992, Ouellet et al. 1996, Stuart-Smith et al. 1997, Rettie and Messier 1998), and the ability to reduce predation risk is expressed at large spatial scales (i.e. through selection and use of peatland complexes). According to recent theory, at finer spatial scales, ecological factors less limiting than predation are reflected in habitat selection (Rettie and Messier 2000). Our work does not fully support these views; habitat selection appears to permeate spatial and temporal scales. While selection favours more lichen-rich habitats at the scales we measured, caribou still preferred peatlands to uplands at

the home range scale, as was found at the scale of the population (Schneider et al. 2000). Temporally, habitat selection did not vary with season, implying that caribou did not select habitats solely on forage availability. While habitat selection is certainly hierarchical (Johnson 1980), selection that is hierarchical in response to limiting factors appears difficult to demonstrate for caribou in northern Alberta.

Regarding management implications of our study, conservation efforts should focus on accounting for the cumulative effects of disturbances, both natural and anthropogenic (McLoughlin et al. 2003). Wildfires reduce the lichen availability for caribou, and should not be overlooked as a measurable disturbance influence on caribou. Our data suggests that caribou avoidance of recent burns is less evident in very heavily burned areas, and caribou may have alternate strategies that allow them to cope effectively with forage losses. It is plausible that woodland caribou have evolved within a wildfire-dominated system that is variable in size, shape, and among years, so spatial patterning and intensity of wildfires may also contribute to unknown trade-offs associated with wildfire occurrence. Mechanisms of caribou habitat selection should be further examined by comparing selection patterns before and after wildfire events and in conjunction with measures of predator and alternative prey density – this would aid in confirming whether selection patterns of caribou are a result of functional responses to habitat availability.

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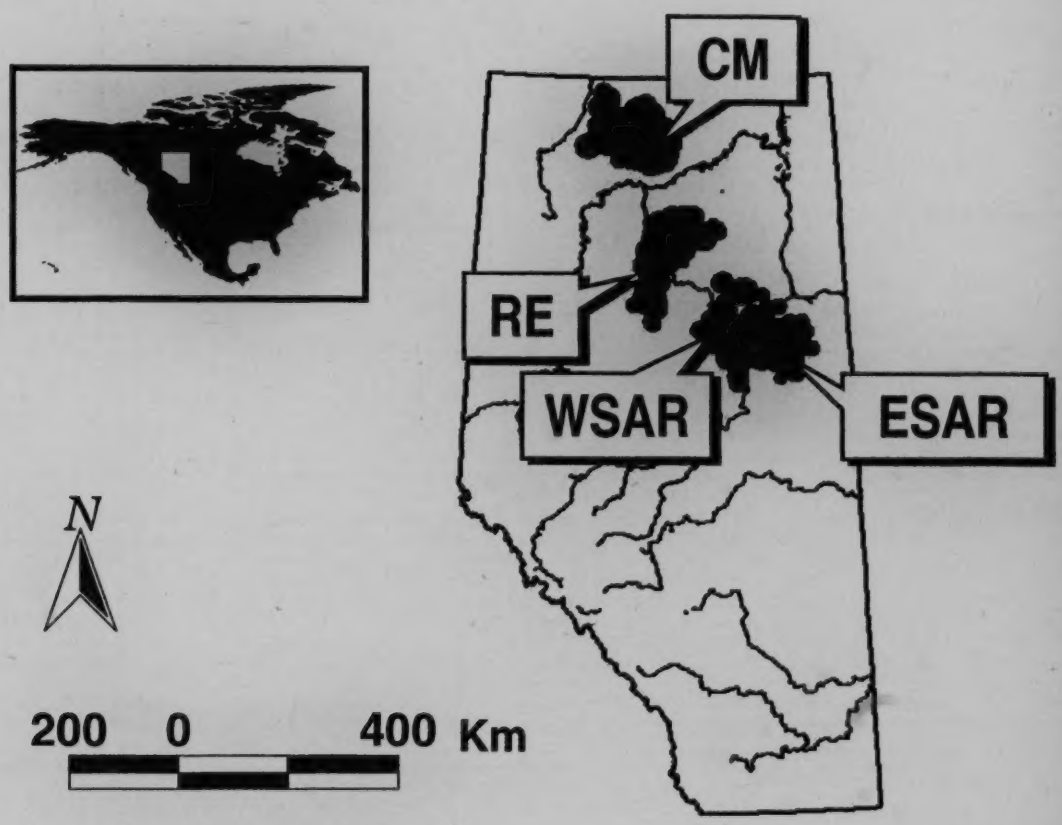

Figure 4.1 Study areas within the province of Alberta, Canada. Regions CM, RE, WSAR, and ESAR refers to the regions Caribou Mountains, Red Earth, West-Side of the Athabasca River, East-Side of the Athabasca River respectively. Major rivers are show in black; inset shows the provincial location within North America.

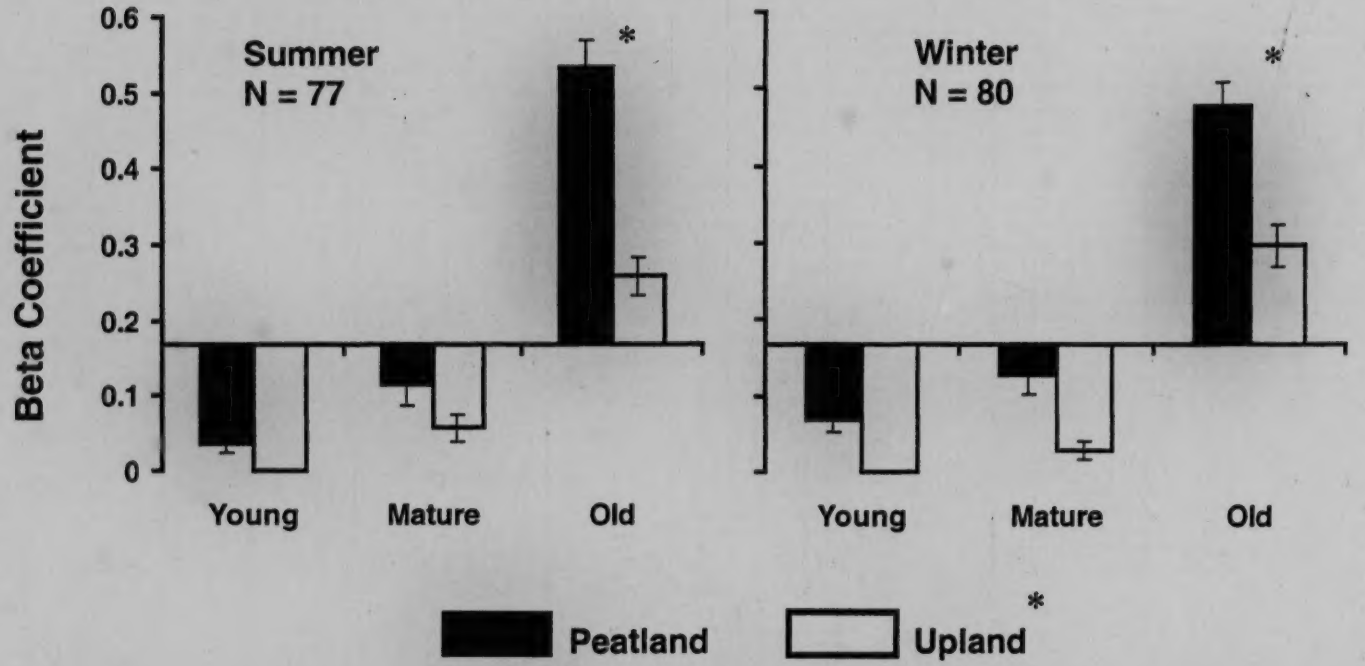

Figure Error! No text of specified style in document..**2** Seasonal caribou selection pattern within the West Side Athabasca River region (WSAR) at the home range scale. Young refers to habitats burned with 10 years; mature represents areas burned within 11 to 50 years, and old refers to areas not burned in at least 51 years. Beta coefficients below the line of random use indicate proportionally lesser use than availability (avoidance), while coefficients above the line imply proportionally greater use than availability (selection). Error bars represent standard error measures, and asterisks indicate significant differences between peatland and upland selection within the corresponding seral stage.

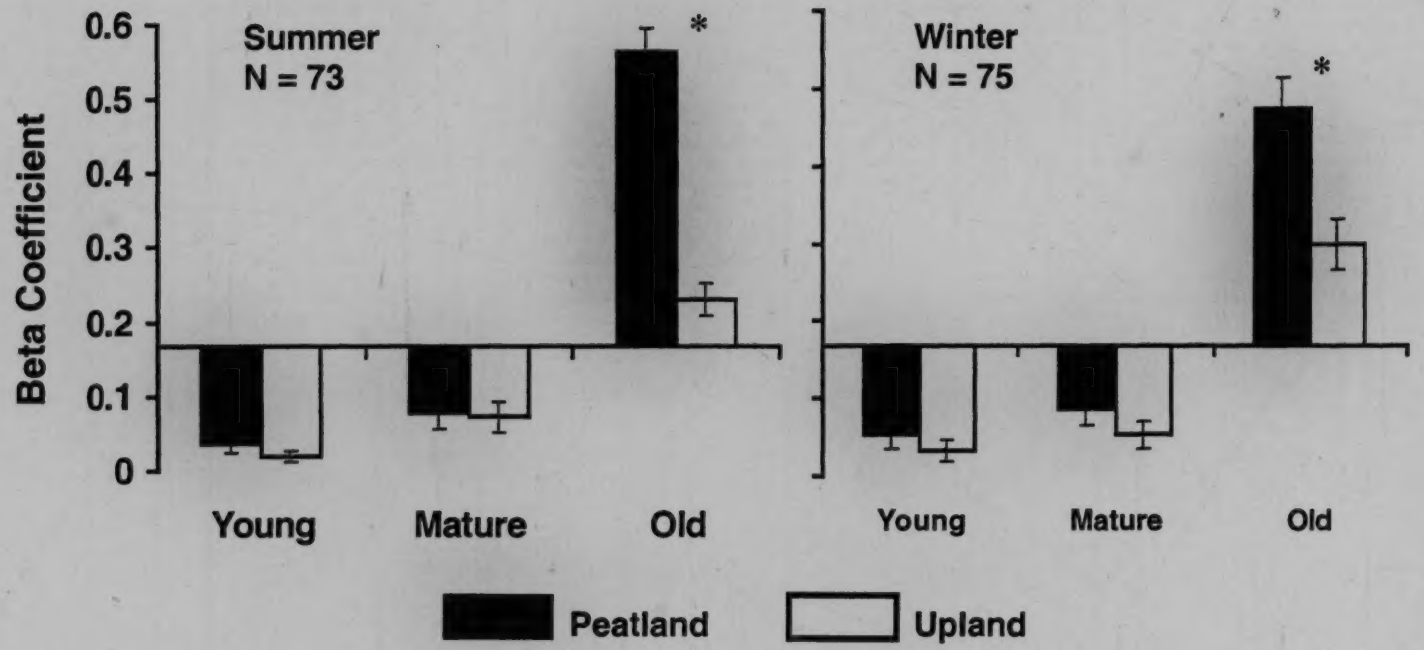

Figure Error! No text of specified style in document..3 Seasonal caribou selection pattern within the East Side Athabasca River region (ESAR) at the home range scale. Seral stages and interpretations of selection and avoidance are identical to those in Figure 4.2.

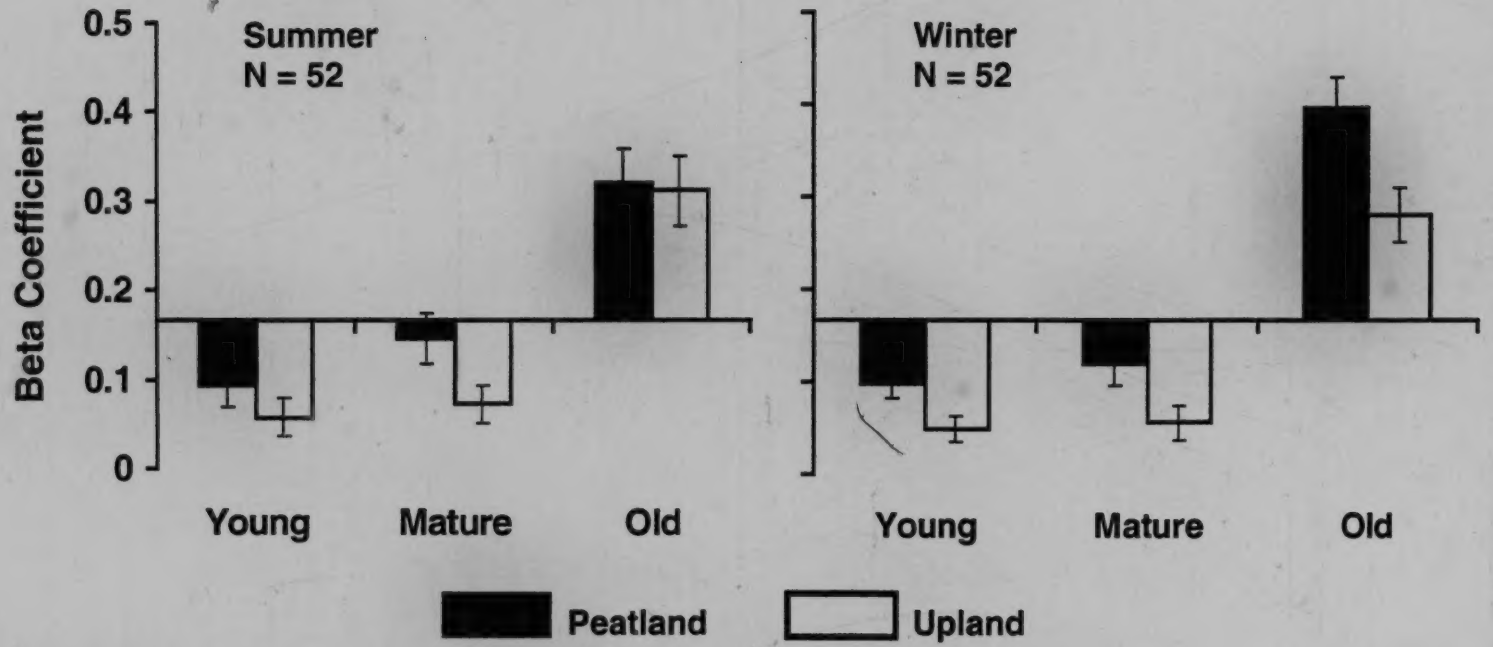

Figure Error! No text of specified style in document..**4** Seasonal caribou selection pattern within the Red Earth region (RE) at the home range scale. Seral stages and interpretations of selection and avoidance are identical to those in Figure 4.2.

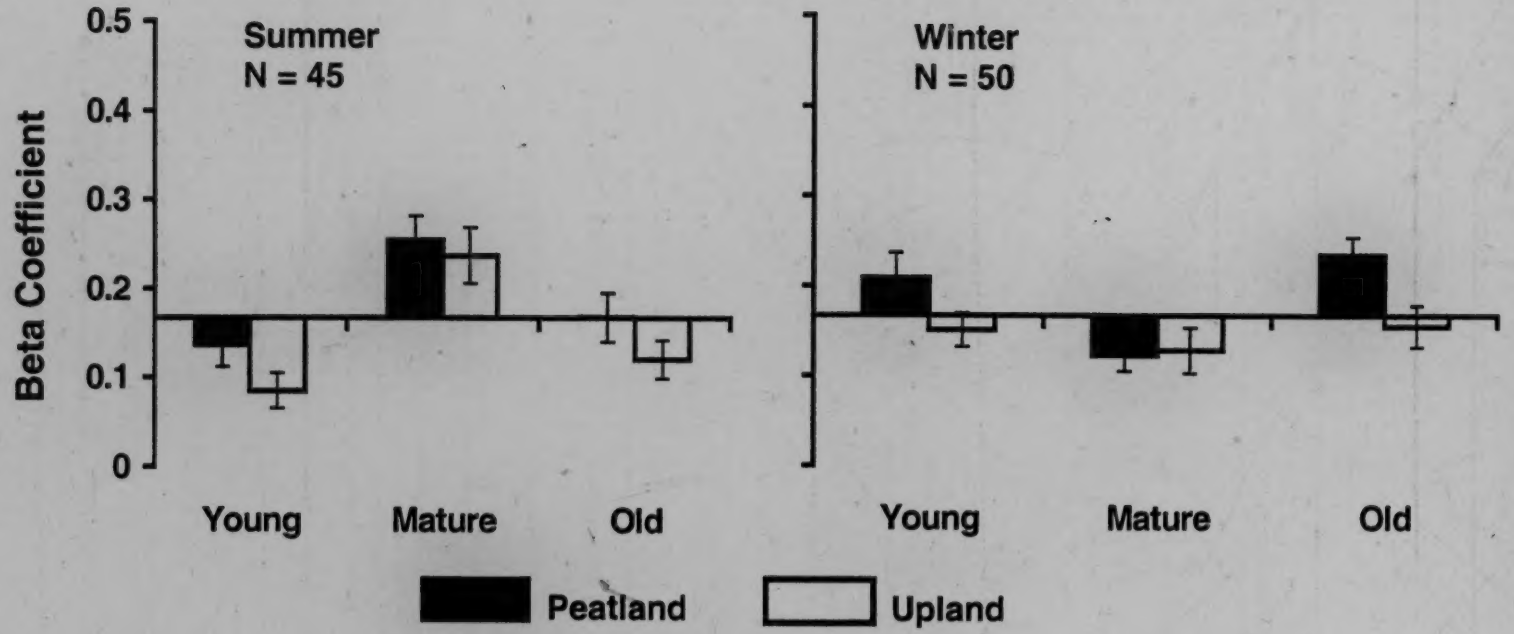

Figure Error! No text of specified style in document..**5** Seasonal caribou selection pattern within the Caribou Mountains region (CM) at the home range scale. Seral stages and interpretations of selection and avoidance are identical to those in Figure 4.2

Table 4.1 Percent home range composition of the 6 different habitat types among the four study regions. Regions CM, RE, WSAR, ESAR, and CLAB refers to the regions Caribou Mountains, Red Earth, West-Side of the Athabasca River, East-Side of the Athabasca River.

Mean proportional home range composition						
	Peatland			*Upland*		
Region	Young	Mature	Old	Young	Mature	Old
WSAR	5.3	2.7	65.6	0.2	2.2	24.2
ESAR	6.0	2.7	52.1	6.7	2.9	29.7
RE	15.6	13.6	16.7	9.4	6.7	38.1
CM	14.3	19.5	26.4	7.3	16.5	16.0

Table 4.2 Percent use of the 6 different habitat types at the home range scale among the four study areas. Regions CM, RE, WSAR, ESAR, and CLAB refers to the regions Caribou Mountains, Red Earth, West-Side of the Athabasca River, East-Side of the Athabasca River.

Mean percent telemetry locations						
	Peatland			*Upland*		
Region	Young	Mature	Old	Young	Mature	Old
WSAR	2.1	2.4	69.8	0.1	1.9	23.6
ESAR	2.1	2.8	53.3	2.5	3.8	35.5
RE	10.3	6.0	23.0	6.0	2.6	52.1
CM	15.6	19.6	22.9	8.7	16.7	16.4

CHAPTER 5. TRENDS IN WOODLAND CARIBOU POPULATIONS AND HABITAT AVAILABILITY IN SCENARIOS OF CUMULATIVE EFFECTS AND CLIMATE CHANGE

Introduction

The role of anthropogenic disturbances in influencing animal mortality, genetic diversity, habitat availability, and population dynamics are well known among many taxa. Unfortunately, the effects of individual disturbance elements on a given species are usually documented in isolation. For example, roads may increase mortality for grizzly bears (Mace et al. 1996, Benn and Herrero 2002). Electrical transmission lines have been shown to kill birds (Manosa and Real 2001), pipelines and seismic lines in boreal forests can decrease the habitat available for woodland caribou (Dyer et al. 2001), and population declines of Northern Spotted Owls have been attributed to logging in the Pacific Northwest (Thomas et al. 1990). Multiple disturbances will continue to take place within finite landscapes, and measures of species responses' to such perturbations will likely form a key component of cumulative effects assessment (CEA). The identification of development thresholds identified from CEA will likely be especially important for species whose populations are in decline and potentially limited by factors associated with multiple land uses.

In western Canada, the boreal forest has experienced unprecedented growth in industrial development over the past century (Schneider 2002). Activities related to oil and gas extraction, timber harvesting, mining, and agriculture are common throughout boreal regions of Alberta. Coinciding with this historical development, woodland caribou in Alberta have been extirpated from southern areas of their range

(Gray 1999), and are now considered 'threatened' in Alberta (Dzus 2001; COSEWIC 2002). Recent work has documented that most populations of caribou in northern Alberta continue to decline (McLoughlin et al. 2003), and habitat alterations by industrial activity has reduced the effective habitat available for caribou (Dyer et al. 2001, 2002) and increased the susceptibility of caribou to predation (James and Stuart-Smith 2000).

Woodland caribou are known to prefer and extensively use peatlands in Alberta (ombotrophic bogs and fens) as part of a strategy that may limit their exposure to predators (Stuart-Smith et al. 1997, James 1999, Schneider 2000). Caribou in boreal Alberta are wide-ranging and non-migratory, occupying annual home ranges over 500 km^2 (Bradshaw et al. 1995; Stuart-smith et al. 1997; Schneider et al. 2000). The wide distribution of caribou and habitat selection strategies leave populations relatively isolated from forest harvesting, but the network of roads associated with forestry operations, wildfires, and disturbances related to petroleum exploration and production activities, all intersect caribou ranges. Natural disturbances appear to have little influence on the distribution of caribou (Dunford 2003), but because certain burned habitat types are avoided by caribou (Dunford 2003), they may play an important role in defining habitat availability for caribou should climatic changes alter the frequency of such disturbances.

Recognizing that land uses and disturbances take place on caribou ranges, industrial developers have undertaken mitigative measures as part of their operation, including reducing the width of seismic lines (a key disturbance resulting from oil and gas exploration), and co-planning of roads among industrial users (Dzus 2001).

Experiments are also underway to identify the best means of reclaiming past seismic disturbances, and industrial operators are required to document how their operations incorporate guidelines for 'best practices.' Such conservation measures however, may be superficial, and effective only in the short term if the magnitude and duration of development in caribou range overwhelms mitigative measures. Lacking in the conservation planning for caribou was an evaluation of the potential for different land use strategies and mitigative measures to influence the declining trend in caribou populations and habitat availability.

ALCES ® (A Landscape Cumulative Effects Simulator, Forem Technologies 2002), a STELLA-based simulation model, provides the means to explore ecological and industrial risks associated with different development trajectories (Schneider 2002; Schneider et al. 2003). We used ALCES ® to examine how cumulative effects from both industrial development and natural disturbances may influence woodland caribou in Alberta over the next century. Specifically, our intent was to (1) establish what possible cumulative effects industrial activity may have on caribou habitat and population trends, (2) examine the relative effectiveness of multi-industry mitigation measures, and (3) examine whether an increase in fire frequency resulting from climate warming would alter trends of caribou habitat availability and populations among development scenarios.

Study Areas

Our study area encompasses 5,963,847 hectares of northeastern Alberta, Canada, that makes up the Forest Management Agreement tenure of Alberta Pacific

Forest Industries Inc. (Figure 5.1). The region is naturally fragmented into poorly drained peatlands in low-lying areas and well-drained uplands and river valleys. Peatlands support black spruce (*Picea mariana*) and larch (*Larix larcina*) trees, and understory vegetation that includes Labrador tea (*Ledum groenlandicum*), sheathed cotton-grass (*Eriophorum vaginatum vaginatum*), and *Sphagnum* mosses (Vitt et al. 1996). Uplands are dominated by stands of white spruce (*Picea glauca*), jack pine (*Pinus banksiana*), and trembling aspen (*Populus tremuloides*), and shrubs such as wild rose (*Rosa acicularis*) and alder (*Alnus crispa*). Topographic variation is minimal in all regions.

Approximately 40% of the study area is composed of merchantable forests; the bulk of which are hardwood stands (Table 5.1). Greater than half of the study area is composed of peatlands that contain little to no merchantable timber. Oil and gas deposits are abundant in the region, as evidenced by the current footprint of seismic lines, wells, pipelines, and roads (Table 5.1). Current reserves for conventional petroleum deposits (i.e., crude oil, natural gas) are being depleted in Alberta, and consequently, reduced growth will be likely among the conventional oil and gas sector in the area (AEUB 2001). However, oil-saturated sands (oil sands) present in the study area have been exploited only minimally. Oil sand reserves in Alberta contain more recoverable petroleum than those of Saudi Arabia, and development of reserves such as those present in the study area are expected to support growth in the energy sector for at least the next 100 years (AEUB 2001).

The study area largely encompasses the ranges of two caribou populations described by McLoughlin et al. (2003; East and West Side Athabasca River

populations). While the East Side population is currently declining, the West Side population is considered stable (McLoughlin et al. 2003). The total number of animals in the study area is unknown, but likely less than the 1000 animals we use in our simulations.

Methods

The ALCES ® Model

We used an aspatial simulation program, ALCES ®, that tracks industrial footprints and associated ecological parameters in scenarios of industrial development. The model functions on a STELLA platform, and is an effective simulation tool for exploring the relative consequences of different land use strategies for strategic-level planning. ALCES ® is based on a user-defined landscape and quantitative assumptions of future industrial activities, natural disturbances, ecological responses to disturbance, and reclamation of disturbed areas, to predict possible landscape compositions in 1-year time steps. Schneider et al. (2003) provides a more detailed description of the model development and utility.

Parameters of industrial activity and growth in the model are largely deterministic, with some exceptions (e.g. forest harvest resulting from random wildfire events). Demographic and habitat availability parameters were also deterministic, but population trends were altered by the stochasticity associated with wildfire (e.g. survival varied with anthropogenic edge density; wildfire could alter edge density due to salvage logging). To account for the stochasticity introduced by wildfire, we ran each scenario 50 times, and used mean estimates of caribou habitat

availability and population size at each time-step to document trends among simulation timelines.

Industrial Growth Parameters

We tested three scenarios of industrial development on the landscape: 'current trends,' 'best practices,' and 'no development' (Table 5.2). The no development scenario had no additional industrial footprints added, but also no active reclamation of the current footprint. Levels of growth in the energy sector follow those presented in Schneider et al. (2003) and model the present growth rate of drilling activity that declines as the conventional oil and natural gas reserves are depleted (see Schneider et al. 2003 for comparisons of model assumptions and historic energy sector activity).

Caribou Habitat Parameters

Our measures of habitat availability in response to industrial and natural disturbances for simulations were based on estimates of 'habitat values' for disturbed habitats (Table 5.3). Estimates of habitat values were produced from a workshop aimed at quantifying woodland caribou habitat requirements and responses to disturbances for ALCES ®; workshop attendees included 20 provincial biologists, university professors, and caribou researchers, and relied on interpretation of published references and expert opinion (Caribou Delphi Workshop 2001). Habitat values can be roughly interpreted as the probability of expected use, ranging from 0 (no expected use) to 1 (100% of expected use), and were used in the model's calculation of habitat availability for caribou. For example, Dyer et al. (2001)

demonstrated that the relative use of areas buffering well pads and seismic lines was 60% of what would be expected from random use, hence the 250 m buffer surrounding well pads in the model had a habitat value of 0.6. Similarly, habitat value for the area surrounding cut-blocks created from forest harvest (Table 5.3) was based on Smith et al.'s (2000) avoidance distances, but we also used a Habitat Value of 0.6 from Dyer et al.'s (2001) avoidance measures surrounding seismic lines and well sites. We estimated the value of habitats within seral stages for both peatlands and non-peatlands, based on analysis of what seral stages of habitats caribou included within their home range.

Caribou Demographic Parameters

Caribou populations in northeastern Alberta and boreal regions of Saskatchewan have been studied since the early 1990's (e.g. Stuart-Smith et al. 1997, Rettie and Messier 1998, McLoughlin et al. 2003); hence, demographic parameters for the local populations of caribou were available for the simulations. Fecundity and survival of all individuals was assumed to be independent of caribou density. We also use previously unpublished data that suggests adult caribou survival may decrease as a function of linear feature density (Figure 5.2).

The model lacks elements of stochasticity and error measures found in Population Viability Analysis (PVA) or Minimum Viable Population (MVP) models (Boyce 1992); demographic parameters herein are largely deterministic and without error measures surrounding parameter estimates. Our intent was not to conduct PVA for the population (and results should not be interpreted as such); rather, our

simulations allow for relative comparisons of population trends among scenarios, and the relative effectiveness of mitigation strategies in influencing such trends.

Climate & Fire Parameters

We attempted to model fire frequency and size distribution following established parameters of fire cycles, and scenarios of amplified fire frequency that may result from climate warming. Because the probability of wildfire occurrence does not appear to be age-dependent (Johnson et al. 1998), the probability of fire events was independent of the time since last disturbance in the model. Wildfire occurrence was described by a random draw from a lognormal distribution in the model; previous data revealed that the extent and frequency of wildfires were well described by this pattern (Armstrong 1999). The distribution corresponded to an average annual burn rate of 1% among all habitat types in the model (Cumming 1997)

Climate warming was assumed to shorten the fire cycle in the model (in addition to altering vegetative regrowth characteristics), resulting in more frequent occurrence of large, stand-replacing wildfire events by 4% over present levels for every 1° increase in average ambient temperature. We assumed that average ambient temperatures would increase in a linear fashion to 5° Celsius greater than present at the end of the 100 year simulation, resulting in a 20% increase in fires over the simulation period. While our numbers are somewhat arbitrary, they reflect the sentiment that more frequent and severe wildfires will result from climate warming in boreal forests (Stocks et al. 1998).

Results

Habitat Availability

The initial habitat availability for all simulations was 39% of the total landscape area, but declined to less than 3% within 25 years, and less than 1% after 100 years in the current trends simulations (Figure 5.3). The decline in habitat available in the current trends model was a 97.5% decline from current levels. Using best practices and reclamation, habitat availability in the simulations remained relatively constant. In the absence of development though, habitat availability increased steadily to over 50% by simulation ends. The availability of habitat between the climate warming and current climate scenarios was identical, and the variability introduced by random wildfire events produced no detectable variation in mean habitat availability in each time step.

Population Trends

The starting population increased initially to >1200 caribou from time step 0 to 1 (a function of calving taking place prior to population reduction that began after the first 12-month time step in the model). Following this initial increase though, the population declined to less than 20 individuals by 50 years in the current trends simulations (98% reduction; Figure 5.4). Caribou populations in the best practices scenario steadily declined to near 230 individuals after 100 years, and 77% decline from current levels. In the absence of development though, caribou populations never fell below initial levels in the simulations, and although populations declined from years 1 to 25, they grew steadily from years 26 to 100, and numbered >1800 at

simulation ends. Population trends among the climate warming and current climate scenarios were identical, and the variability introduced by random wildfire events was almost undetectable; 95% Confidence Limits surrounding mean population totals at each time step were a maximum of 0.02% of the mean.

Discussion

Our modelling exercise indicates that the current approach towards resource extraction in caribou range can have drastic consequences for caribou populations as a result of the cumulative effect of multiple land uses. The future landscapes we projected, based on current trends in industrial land use, suggest that the declining population trend of caribou in Alberta (McLoughlin et al. 2003) will only accelerate, and caribou could be extirpated among ranges with the next half-century. Habitat availability that is less than 3% of current levels (as in our results) is certainly unlikely to support viable caribou populations. The cumulative effects of multiple land uses will certainly have negative consequences for caribou in Alberta; we suggest the extent of those consequences will be severe and relatively acute.

However, the multi-industry mitigation strategy explored provided clear gains in availability of habitat over the current trends model, and caribou populations, although still declining, had the ability to persist with industrial development in our simulations. Additionally, caribou habitat availability remained stable for caribou when we simulated mitigative techniques. Our mitigation strategies included in the model are a result of communication with resource management and research groups (e.g. Boreal Caribou Committee, Integrated Landscape Management group;

University of Alberta, Adaptive Management Experiment Team; University of Alberta), and include operational tactics that are generally feasible (although not without expense) and an aggressive strategy of reclamation of existing and future footprints. Neither the current trends nor best practices scenarios differ in annual resource production, such as Annual Allowable Cut (AAC) or hydrocarbon exploitation. Despite the additional cost to industrial players that would result from use of specific mitigative measures, economic saving could be possible, through shared road construction or reduced fees associated with smaller linear disturbances (Schneider et al. 2003), which might make these measures more palatable to industrial operators.

Our results suggest that land managers can accelerate the preservation of caribou habitat and populations through aggressive reclamation strategies. The scenario of no development highlights the importance of reclamation in meeting conservation goals. Without any future development, our simulations suggest populations could decline in years 1 through 25, even though no new developments were added to the landscape in these years. After year 25 in the simulations though, the existing industrial footprint was largely naturally reclaimed (or revegetated), and population trends turned positive. Based of these results, if a threshold of industrial footprints exists below which caribou populations enter a terminal decline, that threshold may well be greater than the current landscape footprint.

The role of an amplified fire regime that could result from climate warming had no effect on the availability of caribou habitat or population trends among scenarios. Because the variation in habitat availability from the random occurrence of

wildfires was trivial in the simulations, and because an increase in fire frequency failed to alter trends of caribou habitat availability and populations, we conclude that the influence of wildfires was an inconsequential factor in our model outputs. This can likely be attributed to the uniform habitat values among seral stages of preferred caribou habitat (peatlands). Using data from a finer scale of selection to estimate habitat values than we used (e.g. within home ranges; Dunford 2003), wildfires likely would play a more prominent role in increasing the variability of habitat availability. The role of wildfires in population dynamics of caribou in the wild has been largely dismissed (Bergerud 1974), and we suggest while an increased variability of habitat availability may coincide with an increase in fire frequency, the overwhelming effects of development parameters likely are the key determinate of habitat and population trends.

A key element of conservation planning for caribou is an evaluation of the potential for different land use strategies and mitigative measures to influence the declining trend in caribou populations and habitat availability. This work represents a first step at such an assessment, and indicates that mitigative operational tactics, and aggressive reclamation may allow caribou the opportunity to persist the industrial climate of Alberta's boreal forests. However, more robust error measures and stochastic PVA modelling would be valuable in an attempt to evaluate the potential effectiveness of different land use strategies, and test accuracy of the population persistence we demonstrate within a best practices scenario.

Schneider et al. (2002) used ALCES ® extensively to explore possible future landscape composition in the identical area as our study, and provides results on

future industrial footprints associated with similar activity and growth parameters that we used. Their conclusions regarding the benefit of best practices strategies are similar to ours, but also they stress the importance of using a modelling approach as a support system for decision-making. Measures of ecological risks, such as the gains or losses in caribou habitat, should be compared amongst possible management scenarios in a decision making framework, to ensure that the limitations of finite ecosystems are understood, and conflicting objectives can be balanced in management strategies.

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Table 5.1 Initial landscape composition and industrial footprint within the study area.

Feature	Area (ha)
Natural forested landbase	2,384,174
Hardwood stands	1,224,283
Black spruce stands	51,224
Other forest types (mixedwood, pine, etc)	1,108,867
Fens, bogs, marshes	3,166,586
Lentic & lotic systems	197,499
Anthropogenic disturbances	
Seismic lines	41,082
Well site pads	15,516
Pipelines	22,258
Minor roads	48,205
Major roads	11,606
Well site roads	7,346
Oil sands surface mines	5,829
Total landscape area	5,963,847

Table 5.2 Model assumptions for industrial activity and growth among scenarios.

Industry Parameter	Current Trends	Best Practices
Forestry Sector		
Annual Allowable Cut (AAC) hardwoods	2,400,000 m^3	2,400,000 m^3
AAC softwoods	1,000,000 m^3	1,000,000 m^3
Regeneration failure rate (all stand types)	5 %	5 %
In block losses (haul roads, landings)	4 %	2 %
Energy Sector		
Seismic length per well	5 km	5 km
Seismic line width	5 m	1 m
Seismic lifespan	25 years	4 years if > 1 m width 1 year if < 1 m width
Overlap of new seismic with existing lines	10 %	50 %
Well pad size	1.0 ha	0.8 ha
Number wells per pad	1	2
Well pad lifespan (post production)	25 years	Immediate reclamation
Pipeline width	25 m	Initially 25 m, then 5 m reclaimed immediately
Pipeline lifespan	25 years	20 years
Overlap of pipelines with existing lines	10 %	50 %
Transportation network		
New major roads (permanent)	75 $km*year^{-1}$ for the next 50 years	50 $km*year^{-1}$ for the next 50 years
Road harmonization among sectors	10 %	50 %
Well access road lifespan	Permanent	25 years
Minor road lifespan	Permanent	15 years

Table 5.3 Model assumptions for caribou habitat availability and effectiveness.

Variable	Habitat Value	Source
Roads	500m avoidance (60% of random use expected)	Dyer et al. 2000; James and Stuart-Smith 2000; Dyer et al. 2001
Well Sites	250m avoidance (60% of random use expected)	Dyer et al. 2001
Seismic Lines	100m avoidance (60% of random use expected)	Dyer et al. 2001
Pipelines	100m avoidance (60% of random use expected)	
Cutblocks	540m avoidance (60% of random use expected)	Smith et al. 2000
Habitat Value	Black spruce stands = 0.8	Bradshaw et al. 1995,
	Fens and bogs = 0.9	Stuart-Smith et al. 1997,
	Hardwood stands = 0.02	Schneider et al. 2000;
	Mixedwood stands = 0.05	Anderson 1999; Caribou
	Pine stands = 0.4	Delphi Workshop 2001
	White spruce stands = 0.1	
	Lotic areas = 0.1	
	Moss/lichen areas = 0.8	
	Riparian areas = 0.3	
	Vegetated anthropogenic areas = 0.05	
	Non-vegetated anthropogenic = 0	
	Agriculture areas = 0	
Bog/Fen Age	Forests 0-10a (1.0), 20-40a (1.0), 50+a (1.0)	Dunford (unpublished)
Forest Age	Forests 0-10a (0.38), 20-40a (0.57), 50+a (1.0)	

Table 5.4 Model assumptions for caribou demographic parameters.

Variable	Parameter Estimate	Source
Total initial population	1000 Caribou	McLoughlin 2003
Initial age and gender composition	6.4% Juvenile male 2.6% Juvenile female 6.4% Yearling male 2.6% Yearling female 37.6% Adult male 44.4% Adult female	Edmonds 1988; Stuart-Smith et al. 1997
Fecundity	0.9 Calves*adult female^{-1} 0.7 Calves*yearling female^{-1}	Stuart-Smith et al. 1997; Rettie and Messier 1998; McLoughlin et al. 2003
Juvenile Survival	0.22	McLoughlin et al. 2003
Yearling Survival	Variable with linear feature edge density; 0.82 to 0.91 (See Figure 5.2)	James 1999; this paper
Adult Survival	Variable with linear feature edge density; 0.82 to 0.91 (See Figure 5.2)	James 1999; this paper

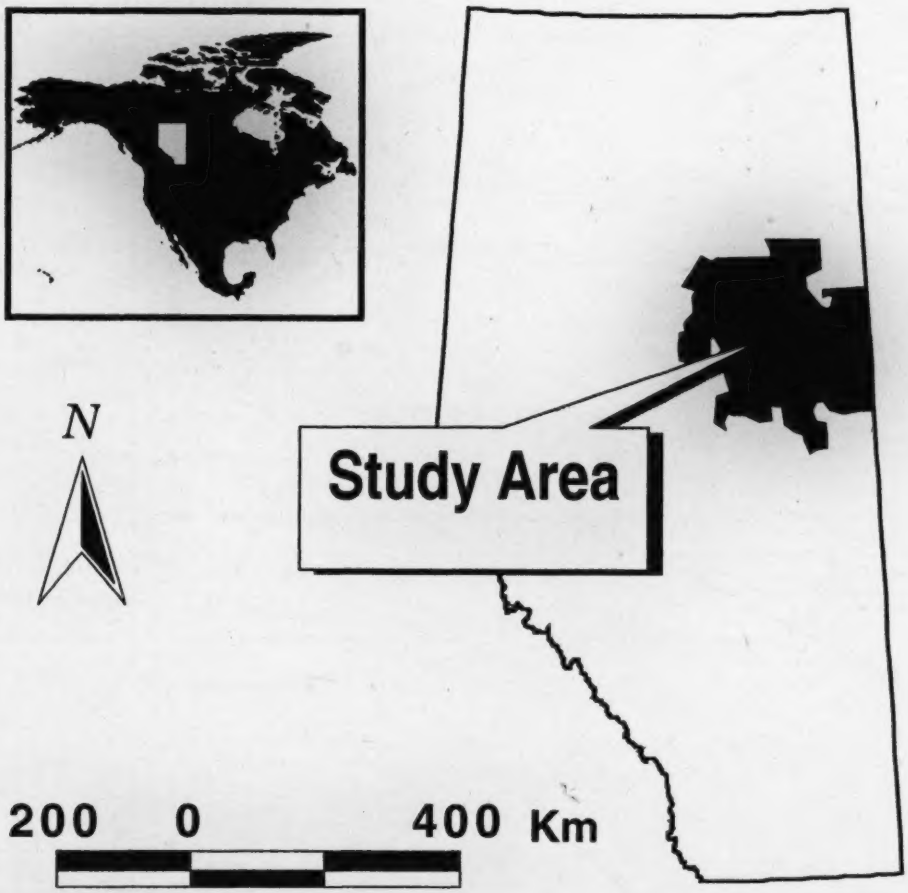

Figure 5.1 Study area (Alberta Pacific Forest Industries Forest Management Agreement) in northeastern Alberta. Inset shows provincial location within North America.

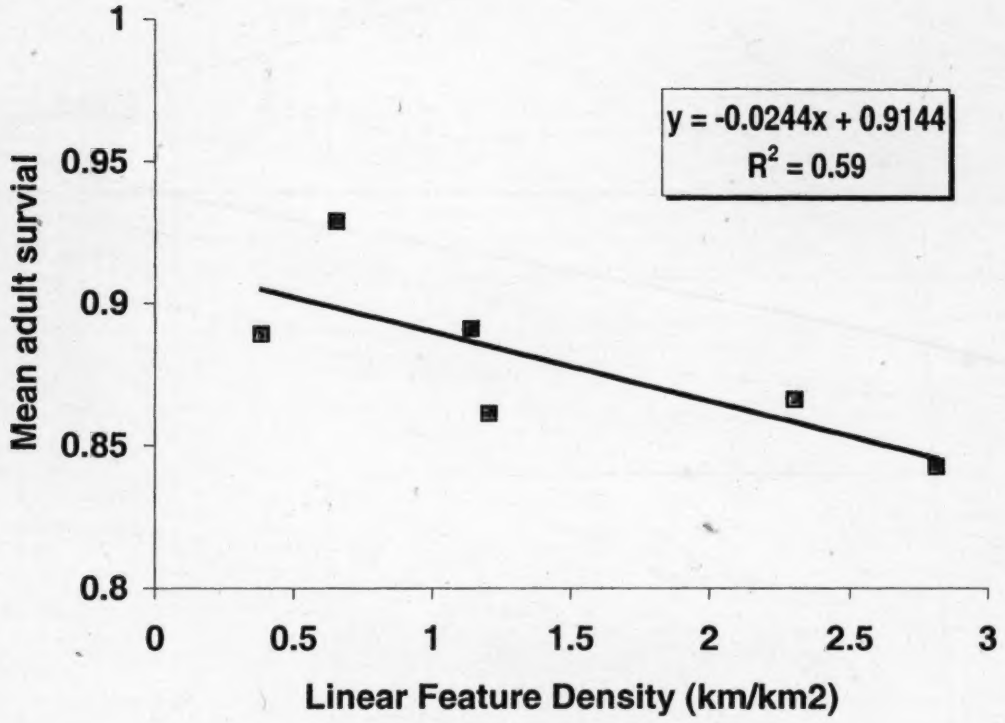

Figure 5.2 Mean adult female caribou survival as a function of linear feature density among six woodland caribou ranges in northern Alberta (Source: Boreal Caribou Committee, unpublished data).

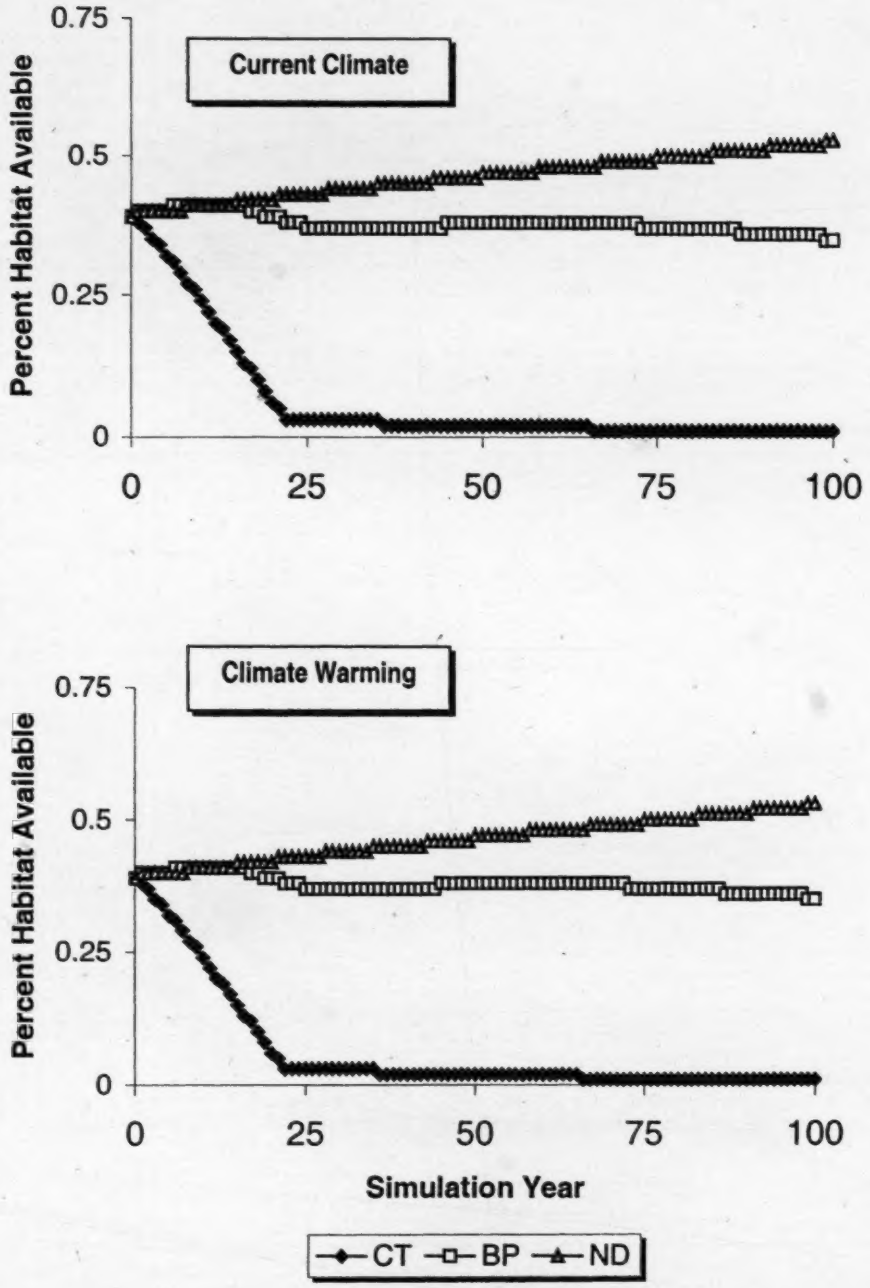

Figure 5.3 Habitat availability for caribou among simulations of current trends in development (CT), best practices and footprint reclamation (BP), and no development or reclamation (ND). Simulations used either current wildfire parameters (Current Climate) or accelerated fire frequency (Climate Warming).

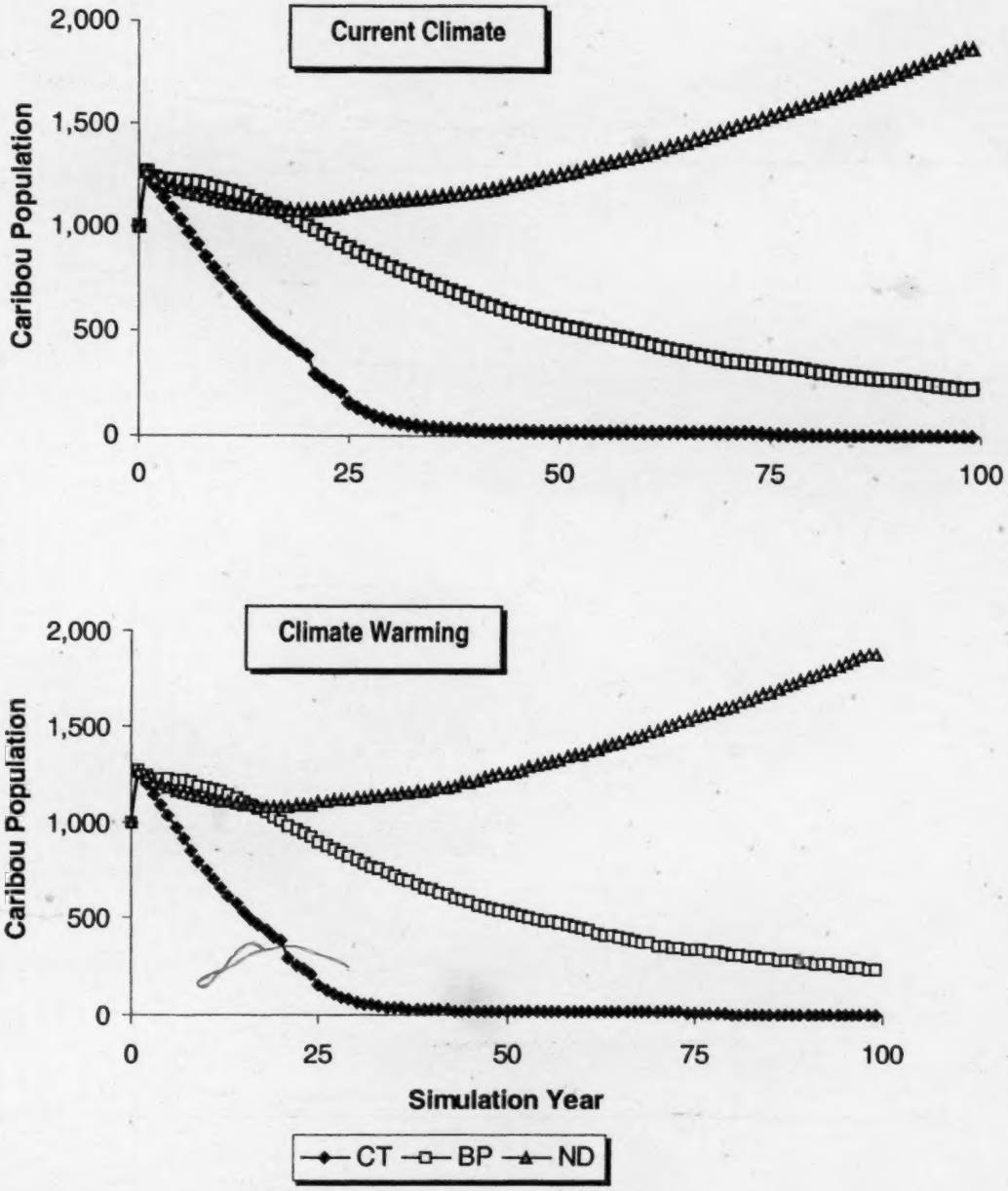

Figure 5.4 Caribou population among simulations of current trends in development (CT), best practices and footprint reclamation (BP), and no development or reclamation (ND). Simulations used either current wildfire parameters (Current Climate) or accelerated fire frequency (Climate Warming).

CHAPTER 6. GENERAL CONCLUSIONS AND IMPLICATIONS

Research Summary & Conclusions

This work focused on documenting the relationship between woodland caribou (*Rangifer tarandus caribou*) and large-scale wildfire events in the boreal forests of northern Alberta. The data presented here confirms that wildfires result in detrimental, but non-permanent alteration of caribou habitats. Distribution and habitat-use behaviour by caribou may allow these animals to cope well with the regular presence of wildfires. Caribou are very widely distributed, so losses of winter forage may be inconsequential if adequate resources are available in other parts of their range. Hence, I detected no significant shifts in annual ranges of caribou following wildfires. Additionally, caribou may be accustomed to an environment where lichens are relatively sparse, and lichen regrowth is relatively quick following fires. In the context of other landscape disturbance elements, the role of wildfires appears secondary in influencing habitat availability or population trends. I conclude that although wildfires are detrimental in terms of winter forage availability, caribou behaviour may allow individuals to compensate for such disturbances.

Management Recommendations

The management of caribou in Alberta is certainly at a 'crossroads' (Schneider 2002). Managers in Alberta now have the benefit of over a decade of detailed telemetry and population monitoring data for over 300 caribou, accurate measures of the responses of caribou to noise (Bradshaw et al. 1995), forestry activities (Smith et al. 2000), oil and gas exploration and production (Dyer et al.

2001, 2002, James and Stuart-Smith 2000), and roads (Dyer 2001, 2002). The response of caribou to wildfires are described herein, and digital spatial data is easily available that documents wildfire occurrence over the past 75 years (Alberta Sustainable Resource Development 2003). We are also able to model the frequency and probably of future wildfire events (Cumming 1997, Armstrong 1999) that may impact caribou. The habitat types used and selected by caribou are well understood (Bradshaw et al. 1995, Stuart-Smith et al. 1997, Anderson 1999, Schneider 2000), and landscape characteristics affecting the survival of caribou have been identified (Dunford et al. *In Review*). Further, the advent of the sophisticated modelling techniques has also allowed us to strategically examine the effectiveness of our conservation measures in the context of 'meaningful time' (ALCES ®; Forem Technologies Ltd. 2002). There can be no argument that we have insufficient information to make the necessary decisions to conserve caribou in Alberta; indeed, the data available for caribou managers in Alberta is certainly among the best wildlife management dataset in the world. The challenge therefore, is to use this data to develop and implement an effective strategy for the conservation of caribou.

Balancing the societal benefits of development in boreal regions with caribou conservation initiatives is difficult. Thus far, conservation strategies for caribou have relied on the voluntarily mitigative steps by Industry, guidelines for developers, and a moratorium on caribou harvest. The role of protected areas has been limited in conservation strategies. Recovery planning underway should explicitly address these measures, and question their merit. I suggest that an emphasis on limiting development in sensitive areas, reclaiming past human footprints, and protecting key

areas should be the foci of future recovery plans. Admittedly, the success (or failure) of the current recovery planning exercise may not be apparent for decades. Therefore, monitoring of populations and trends should continue, as they are key indicators of whether implemented management strategies are effective, or if they require modifications.

In areas where large wildfires have occurred, management should ensure that caribou conservation strategies are congruent with the species' ecology. The wide-ranging behaviour of caribou may be a means of coping with forage losses, so managers should ensure planning units are large enough to accompany the needs of caribou in the event of wildfire (i.e., >10,000 km^2). Additionally, future development planning should include recent wildfires, and perhaps the likelihood of wildfires occurring, as a part of Cumulative Effects Assessment for a given planning unit. Large scale 'zoning' may provide some means of conserving large portions of caribou range, while developing resource-rich areas to meet the needs of society.

The role of protected areas has been overlooked in northern Alberta, perhaps because it is simply too late to protect wildlife habitats in some areas, or perhaps because policy makers are hesitant to inhibit development within portions of Alberta. Protected areas may be a useful component of a caribou conservation strategy and recovery plan. As part of a zoning strategy, permanent preservation zones would be beneficial in ensuring that the genetic diversity of caribou could be adequately maintained, and as a safety factor should stochastic occurrences depopulate other regions. Protected areas need not exclude development absolutely, and industrial activities might occur at varying intensities within planning units or protected areas.

If thresholds for development exist beyond which caribou cannot maintain their populations, we should recognize that they might already have been breached in parts of northeastern Alberta. In such areas, reclamation of existing features should be the focus of managers and policymakers. This will ensure that if caribou are extirpated within a given area, the restoration of populations may be possible in the future. Activities such as translocation of caribou and predator depopulation may also be required to prevent the loss of local populations. Those activities are expensive and invasive though, and managers may be faced with tough decisions regarding the benefit of such a program. Because caribou populations continue to decline throughout most of the province (McLoughlin et al. 2003), the possibility of local extirpations is likely, and managers should be preparing a strategy to cope with local population crashes.

I suggest that the biggest impediment to successful conservation planning and action in Alberta lies in the regulatory structure and tenure of resource allocation in Alberta. The institution of resource development policies has reinforced a premise of continued industrial growth in the boreal forest, largely ignorant of caribou populations in Alberta over the past century. Conservation planning for caribou cannot be independent of large-scale landscape altering processes (including wildfire) to be successful. The current practice of piecemeal mitigative measures for caribou amounts to merely re-arranging deck chairs on an ecological titanic, and is certain to fail. Data presented in this work suggests that natural disturbance elements have a minor role in influencing the space-use and population dynamics of caribou, and thus,

addressing the anthropogenic factors implicated in caribou declines will have far greater success in preserving caribou in Alberta.

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